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# Reef Fish Assemblages on Near-Shore Coral Reefs: The Effects of Habitat Structure, Degradation and Rehabilitation

Thesis submitted by

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James Cook University of North Queensland

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# ABSTRACT

The literature on the ecology of coral reef fishes is divided over the importance of habitat structure and resource availability in determining the characteristics of fish assemblages. The profile of this issue has increased, as a consequence of the increasing degradation of coral reef habitats and the need to investigate active habitat restoration as a means of reversing this process. This thesis investigates emerging generalisations about the relationships between fish communities and the characteristics of coral reef habitats on near-shore reefs, by focussing on two widely separated geographic locations (Phuket, Thailand and Orpheus Island, GBR), spatial and temporal patterns within these locations, and a range of common fish taxa (including Pomacentridae, Labridae and Chaetodontidae). The thesis employs a wide range of observational tools, including multivariate analysis to detect and describe spatial and temporal pattern in fish-habitat associations. It culminates in a series of experimental manipulations, including degradation and restoration, to test cause-effect links between fishes and different components of their habitat.

The first field programme (Chapter 3) was designed to develop an integrated transect sampling protocol for estimating fish abundance and the cover of benthic assemblages on inshore reefs. Techniques were employed to optimise sampling, not only for fish and benthic organisms separately, but also for detecting relationships between them. The performance of sampling using different transect lengths and widths was evaluated at two locations and two habitats within locations, in terms of absolute estimates, precision and efficiency. The influence of transect width on fish abundance estimates varied across localities and habitat. In contrast, transect length did not show a significant effect on estimates of abundance and shorter transects were more efficient to obtain desired levels of precision. For measuring habitat structure, two sampling techniques, Fixed Density Point (FDP) transects (in which cover was estimated using a fixed number of semi-random points) and Line Intercept (LIT) transects (in which cover was measured in terms of distance along a tape), were compared for transects of different length. The FDP method was more efficient than LIT, in terms of precision, but FDP underestimated the cover of less common habitat categories, particularly on short transects. Averaged across species, habitats and localities, a 30 m transect was considered the most efficient length for both sampling techniques. Examination of the effect of

different sampling techniques and transect dimensions on the observed patterns in fish-habitat relationships indicated that only sampling with high precision (for both fish and benthos) provided consistent results. To achieve this goal, it was calculated that six replicates of a 5 x 30 $m^2$ transect were necessary for fishes and six replicates of 30 m - LIT were necessary to adequately describe habitat structure. To examine relationships between the two, fish and benthic organisms were quantified using the same transect lines.

At Phuket, Thailand, the relationship of habitat structure with coral reef fish assemblages from three families; Labridae, Chaetodontidae and Pomacentridae, were investigated during 1994 and 1995 (Chapter 4). A variety of linear and parabolic relationships between living coral cover and community parameters were detected, by comparing multiple locations. Species richness was maximal at intermediate coral cover and evenness was maximal at the extremes of coral cover. Canonical Correlation Analysis identified family-wide spatial associations between fish and benthic habitat variables. In statistical terms, the Chaetodontidae were responsive (their presence/abundance depend on habitat structure variables), the Labridae were predictive (their presence/abundance indicate habitat structure variables) and the numerically dominant Pomacentridae were both responsive and predictive. Temporal variation in habitat structure, including reef degradation and unassisted recovery also influenced the composition of fish communities in predictable ways.

In the third field study (Chapter 5), the relationships between wrasse assemblage and habitat structure were investigated on fringing reefs of 3 inshore islands of the central GBR (Dunk, Orpheus and Magnetic Islands) during 1993 and 1994. Some linear and quadratic the relationships between % cover of major benthic life-forms and community parameters were detected. Living coral and algae appeared to have negative relationships with wrasse assemblages (abundance, diversity), while they were positively related to the cover of dead and/or soft coral. Canonical Correlation Analysis demonstrated significant relationships between multivariate descriptors of both fish assemblage and habitat structure. The nature of these relationships tended to be predictive for habitat structure and responsive for the fish. Temporal patterns in the structure of the wrasse assemblage were studied over a two year period at Orpheus Island. The community parameters indicated some variation at a seasonal scale, but stability over an annual scale. Canonical Discriminant Analysis indicated that, despite temporal fluctuations, spatial patterns in the structure of labrid communities persist over time.

The effects of habitat degradation on coral reef fish assemblages were experimentally investigated at two locations on Maiton Island (Phuket, Thailand - a degraded reef) and two locations on Orpheus Island (GBR, Australia - a relatively "pristine" reef) (Chapter 6). The familial structure of fish assemblages differed between these two regions, with Maiton Island co-dominated by pomacentrids and labrids, while pomacentrids were dominant on Orpheus Island. In general, fish responded negatively to living hard coral degradation, showing decreased diversity, species richness and abundance, and predictable declines in coral-associated species. The magnitude and details of the response were specific to study areas, the pre-existing condition of the habitat and taxonomic group. In contrast to hard corals, the removal of soft coral appeared to have a positive effect to most fish, apparently because domination of habitat by soft corals reduces habitat complexity.

Experimental rehabilitation of biotic habitat types was also carried out at the two geographic locations (Maiton and Orpheus Islands) to assess whether habitat rehabilitation alone was sufficient to restore fish assemblages (Chapter 7). The experiment was set up by introducing focal habitat structures (branching coral, massive corals and soft corals) to patches of degraded, largely dead coral reef. Branching *Acropora* induced the greatest changes in fish communities at most locations, particularly Orpheus island, where there was a consistent increase in the diversity and abundance of pomacentrids. The smaller and more site-specific effects at Maiton Island may be a consequence of the greater habitat degradation in this region, hence, restoration may be slower to act. The re-introduction of massive corals and soft corals demonstrated fewer effects, reflecting the lower physical complexity of these substrata.

The knowledge gained from this study was applied to develop a decision tool for coral reef resource management (Chapter 8). Management decisions for coral reef systems are often made on the basis of limited biological data, and status assessments are often over-simplistic, being based simply on total hard coral cover. A new assessment procedure is proposed based on a hierarchy of different levels of data availability. Four primary level indices were developed based on reef development and different components of benthic assemblage. The four indices were the Development Index (which adjusts coral cover estimates on the basis of reef development), the

Condition Index (which adjusts coral cover estimates on the basis of coral assemblage condition), the Algal Index (which adjusts algal cover estimates on the basis of its potential to occupy non-living coral area), and the "Other fauna" index (which adjusts other fauna cover estimates on the basis of their potential to occupy non-living coral area). A secondary level index was also developed by integrating Development and Condition Indices, using a multi-dimensional ranking method. The application of this new procedure was carried out in three geographical regions: the east of the Gulf of Thailand, Phuket and central Great Barrier Reef. The final result is a quality rank for each study site which can be used to set up a priority list for management.

The results of this thesis support conclusions that the biotic habitat structure of coral reefs can have a major influence on the diversity and composition of reef fish communities, which parallels other marine habitats. Both natural and anthropogenic disturbances that disrupt habitats will have major secondary influences on fishes, but if necessary, at least on a small-scale, habitat-rehabilitation may reverse these impacts. The key role of habitat means that it can be developed into indices of reef condition that will also apply to fish assemblages.

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# STATEMENT ON SOURCES
# DECLARATION

I declare that this thesis is my own work and has not been submitted in any form for another degree or diploma at any university or other institution of tertiary education. Information derived from the published or unpublished work of others has been acknowledged in the text and a list of references is given.

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*(Vipoosit Manthachitra)* *(Date)*

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# CHAPTER 1

# GENERAL INTRODUCTION

Ecological studies are fundamentally concerned with the relationships between organisms and their environment (Ehrlich & Roughgarden, 1987; Begon *et al.*, 1996). An important part of the environment is the physical and biological structure of their habitat, which can have a profound influence on the distribution and abundance of species (McCoy & Bell, 1991). These interactions may arise through a number of different mechanisms. Habitat structure may have direct effects on abundance by influencing demographic processes such as recruitment or mortality (Jones, 1988). Behavioural mechanisms such as habitat selection may be a primary determinant of patterns of distribution (Krebs, 1985; Eckert, 1985a; Jones, 1991). By providing refuges, habitat structure may mediate the effects of biological interactions, such as competition and predation (Hixon & Menge, 1991). Other processes, such as disturbance may indirectly affect organisms by modifying the structure of their habitat species (Sano *et al.*, 1987). Community-level patterns, such as species diversity, may also be related to the structure of habitats, with more complex habitats supporting more species (Kohn, 1967; Recher, 1969). However, despite the potentially fundamental role of organism-habitat interactions in determining the structure and dynamics of populations and communities, it has not always been a central part of ecological theory or a popular focus of empirical research (McCoy & Bell, 1991).

The fishes of coral reefs live closely associated with a habitat that is biologically and structurally complex. The habitat itself, as indicated by changes in coral community structure, may be highly variable on a number of spatial and temporal scales (Done, 1982). Distinct communities of reef fish are associated with particular habitat zones on the reefs (Clarke, 1977; Sale & Dybdahl, 1978; Russ, 1984a, 1988b; Meekan *et al.*, 1995; Green 1996). Major fluctuations in habitat structure may result from events such as crown-of-thorns outbreaks (Sano *et al.*, 1984, 1987; Moran, 1986; Faure, 1989) and cyclonic storms (Highsmith *et al.*, 1980; Harmelin-Vivien & Laboute, 1986). Despite this, little is known about the relationships by which habitat structure influencing assemblage structure of reef fish (Jones, 1991), and habitat structure has not played a major role in the development of ecological theory for this group of organisms (see Sale 1991a).

Most of the attention on coral reef fish ecologists has been directed towards particular demographic processes, such as recruitment (Victor, 1983; Sale *et al.*, 1984; Eckert, 1984; Doherty & Williams, 1988; Fowler *et al.*, 1992), or particular ecological interactions such as competition (Jones, 1987, 1988; Clarke, 1989) or predation (Hixon & Menge, 1991; Hixon & Beets, 1993; Caley, 1993). Habitat structure may be implicated in all of these processes. The influence may appear first at the time of settlement, as there is much evidence that fish usually settle onto particular types of habitat or substrata (Sale *et al.*, 1984; Eckert, 1985a, b; Sweatman, 1988; Booth & Beretta, 1994; Green, 1994). Jones (1991) has drawn an attention to the role of habitat structure, in modifying post- recruitment processes such as predation and competition. Habitat structure has been considered as structural factor that may affect the outcome of biological interactions, determining winners or losers in competitive situations, or whether predators consume prey (Jones, 1988; Hixon & Beets, 1989, 1993). However, there are few generalizations concerning which features of coral reef habitats influence the structure of associated fish communities.

As anthropogenic influences on marine habitats such as coral reefs become more widespread, it is becoming increasing important that we determine the strength of linkage between fish and their habitat. Many consider that the coral reefs of the world are in a state of decline (Brown, 1987; Done, 1992; Wilkinson *et al.*, 1993). The effects of such a decline on fish communities cannot be predicted from current theory. Many of the most threatened reefs are near-shore reefs, that are subject to damage through sedimentation, nutrient enrichment and the mechanical disturbance associated with overfishing (Russ & Alcala, 1989; Brown *et al.*, 1990; Grigg, 1994, 1995; McClanahan, 1994). The effects of disturbance and habitat structure on such fish communities have rarely been investigated. While proposals to restore damaged coral habitats have been proposed (Harriott & Fisk, 1988a; Yap *et al.*, 1992; Clark & Edwards, 1994, 1995), whether or not this is likely to lead to a recovery of fish communities is unknown.

This thesis will focus on the spatial and temporal patterns in relationships between habitat structure and coral reef fish assemblages. It focuses on inshore reefs in two different geographic regions, the relatively pristine Great Barrier Reef, and the more impacted reefs of Phuket, Thailand. Observational procedures will be used to examine the spatial relationships between fish assemblages and different aspects of the coral reef habitat. Experimental approaches will explore the effects of disturbance to habitat and the restoration of habitat in the two regions.

## 1.1 OBSERVING RELATIONSHIPS BETWEEN CORAL REEF FISH ASSEMBLAGES AND HABITAT STRUCTURE

Habitat structure can be defined as a particular type of structure that is provided by the arrangement of objects in space (McCoy & Bell, 1991). McCoy & Bell (1991) defined three components of habitat structure: (1) "heterogeneity" - relative abundance of different structure components; (2) "complexity" - absolute abundance of different structure components; and (3) "scale" - the size of the area or volume used to measure heterogeneity and complexity. On coral reefs, the benthic reef assemblage itself is the most obvious component of habitat structure. The coral reef habitat, essentially built up from the growth of living organisms, is essential to maintain ecological integrity of the coral reef ecosystem (Toth, 1995). While both live and dead coral surfaces provide habitat for fishes, these differ in their heterogeneity and complexity. For reefs, habitat heterogeneity includes such variables as percentage area cover and diversity of different coral or patch types (Roberts & Ormond, 1987). Measures of habitat complexity include surface indices (e.g. Dahl, 1973) or rugosity (e.g. Luckhurst & Luckhurst, 1978), and vertical relief (e.g. Luckhurst & Luckhurst, 1978). However, as there is a tendency for a significant correlation between heterogeneity and complexity (McCormick, 1994), it may be difficult to ascribe an effect of one or the other, when determining the influence of habitat on fish communities (Dahl, 1973; Roberts & Ormond, 1987).

Most published studies have examined coral reef fish-habitat relationships by considering the correlation between community parameters of fish with the aerial cover of major benthic lifeforms, particularly living coral (Bell & Galzin, 1984; Roberts & Ormond, 1987; McClanahan, 1994; Green, 1996) or correlation with topographic complexity (Luckhurst & Luckhurst, 1978; Thresher, 1983; Roberts & Ormond, 1987). In general, they have focused on how the structure of a fish assemblage can be predicted from information about habitat structure.

At community level, most of the studies on fish-habitat relationships have been based solely on quantitative observational data that has been collected over a range of spatial scales of sampling (Roberts *et al.*, 1992; McCormick, 1994, 1995; McClanahan, 1994; Green, 1996). When considering large scale gradients, such as patterns across the Great Barrier Reef, concordant changes in the community structure of coral (Done, 1982) and fish assemblages (Williams, 1982) are likely, although associations have not been directly examined at this scale. Large-scale fish-habitat associations have also been reported for the Red Sea (Roberts *et al.*, 1992). Within reef systems, various studies have examined relationships between fishes and features of the habitat, with some emphasizing the importance of particular habitat characteristics in determining community structure (e.g. Bell & Galzin, 1984; Roberts & Ormond, 1987; Bouchon-Navaro & Bouchon, 1989; McCormick, 1994, 1995) and other studies reporting much more loose fish-habitat associations (e.g. Green, 1996). Few generalisations have yet to emerge from these descriptive studies, as the range of taxonomic groups and locations examined is still remarkably small.

One reason for differences in the conclusions of workers examining fish-habitat relationships on coral reefs may be due to different scales and methods of sampling. Different sampling schemes can yield different results even when describing the same system (e.g. Roberts & Ormond, 1987). More attention should be given to establishing reliable sampling techniques and the optimal allocation of sampling effort for both fish and habitat measurements. At most, ecologists will optimize their sampling for focal organisms using established procedures (Pringle, 1984; Andrew & Mapstone, 1987; Bros & Cowell, 1987), but ignore the problems with collecting unreliable data on habitat structure and how this might affect estimates of quantitative relationships between fish and habitat variables. In Chapter 3 , I investigate the influence of different sampling schemes on the quantitative relationships between coral reef benthic and fish assemblages within habitats. The optimal sampling scheme for measuring fish-habitat relationships on larger spatial scales will also be examined.

Biologically important fish-habitat interactions may be species-specific, and different components of the benthic habitat may be more important than others in determining the observed effects. No one measurement of fish community structure may be appropriate for measuring the "fish" side of the relationship, just as a simple description of hard coral cover alone, may be inadequate to quantify the "habitat" side (e.g. Bell & Galzin, 1984; Roberts & Ormond, 1987; Bouchon-Navaro & Bouchon, 1989; Roberts *et al.*, 1992; McClanahan, 1994; Green, 1996). The different choice variables measured in the different studies may be the source of much of the apparently conflicting information. In Chapter's 4 and 5, I use a multi-variable analytical approach to investigate the quantitative relationships between fish and habitats for observations made at a broad spatial scale, from two different geographical regions, the Palm island Group on the central GBR, and near Phuket, Thailand. The temporal stability of the fish-habitat relationships will also be investigated.

## 1.2 ESTABLISHING EFFECTS OF HABITAT STRUCTURE - THE ROLE OF EXPERIMENTS

Experimental manipulation in the field is another more definitive approach to demonstrate the cause-effect linkages between particular features of the habitat and the structure of fish assemblages. Of necessity, these are often limited in scale. Most of the previous experimental studies have been conducted in on small, isolated patches. These have been used to investigate at both population (Doherty, 1982; Jones, 1988) and community level processes (Sale & Dybdahl, 1975; Sano *et al.*, 1984; Hixon & Beets, 1989, 1993), but are subject to the criticism that results do not necessarily help resolve questions on large scales that may be of more human interest (Doherty 1991). Doherty (1991) has suggested that "natural experiments" or large-scale habitat disturbances will provide a valuable opportunity to understand large-scale system dynamics. However, the problems for natural experiment are that they rely on unpredictable events and, in terms of experimental logic, have no appropriate controls, making it difficult to design adequate studies.

Large-scale natural disturbance events, e.g. crown of thorns breakout, storm damage and floods, will clearly be a good opportunity to help resolve the importance of fish-habitat interactions, but may not reveal the mechanisms. These disturbances may represent the direct impact on the fish community (Lassig, 1983) or may be the indirect result of habitat destruction (Kaufman, 1983). Effects have been detected in some studies but not others. For example, Wellington & Victor (1985) could not detect an increase in the abundance of herbivorous damselfish in the Caribbean following the mass mortality of coral and increase in algal cover associated with an El Nino event. In contrast, Sano *et al.*, (1987) found effects of destruction of hermatypic corals by *Acanthaster planci* infestation on reef fish communities. Significant correlation do not represent cause-effect links, just as the absence of correlation do not rule out more complex cause-effect scenarios, when many factors are operating on a system.

Experimental manipulation will continue to be a valuable tool in establishing the importance of different processes, but clearly they need to be expanded to other reef habitats and carried out on larger spatial scales (Jones 1991). The prime advantages of experimental manipulations in the test the effects of disturbance are that the investigator can determine the nature and scale of the disturbance under properly controlled conditions, and can determine the timing and time-scale of the investigation. Jones (1991) pointed out that the impact of disturbance on reef fish assemblages depends on it frequency and magnitude relative to the longevity of the species. Spatial and temporal variation appear to have a great influence on the results of field experiments. The results at one place and one time may not the same with other place and other time (Jones, 1988). In Chapter's 6 and 7 , I use experimental manipulations to examine the effects of disturbance and rehabilitation of contiguous reef habitat on the structure of associated fish assemblages.

## 1.3 DISTURBANCE TO CORAL REEFS AND EFFECTS ON ASSOCIATED FISH ASSEMBLAGES

Coral reef ecosystems appear to be subject to frequent disturbances, from the local damage to individual corals to catastrophic events that result in the destruction of vast areas (Sorokin, 1993; Jackson, 1996). The causes include a wide range of natural events, including storms of different severity, warming of water during El Nino years, extreme low tides, outbreaks of *Acanthaster* and diseases (Highsmith *et al.*, 1980; Brown, 1987; Glynn, 1988; Faure, 1989; Brown & Suharsono, 1990; Glynn & D'Croz, 1990; Brown *et al.*, 1993). To this background, a wide range of anthropogenic impacts are being documented, such as coastal enrichment, sedimentation, overfishing, water discharges, coastal construction, effects of tourism and marine mining, to name just a few (Woodland & Hooper, 1975; Pastorok & Bilyard, 1985; Liddle & Kay, 1987; Russ & Alcala, 1989; Brown *et. al.*, 1990; Neil, 1990; Roberts & Polunin, 1991; Sorokin, 1993; Grigg, 1994, 1995). Natural and anthropogenic influences may interact to have a major influence on the structure of coral reef habitats (Hughes, 1994). The cumulative effects of natural and anthropogenic disturbances, and interactions among these processes, is a major concern for coral reef ecosystems (Sorokin, 1993).

The process of disturbance appears to play a central role in influencing the structure of coral communities, being implicated in explanations of spatial and temporal pattern in species diversity (Connell, 1978; Huston, 1979). These predictions have not been extended to the associated coral reef fish assemblages. Empirical studies show that the degradation of coral reef habitat often has an impact on fish, but the magnitude of the effect depends upon the group of fish, the source of disturbance and its intensity (Kaufman, 1983; Sano *et al.*, 1984, 1987; Williams, 1986; Russ & Alcala, 1989; McClanahan, 1994). No attempts have been made construct predictive models of the effects of disturbance, based on these observations. Disturbances producing mechanical damage to corals, or those contributing to a phase shift from coral to algal domination (see Hughes, 1994) are likely to lead to predictable changes in fish assemblages. Changes in fish community structure can also alter the habitat structure. However, more observational and experimental information, from a range of fish taxa and geographic locations, will be necessary to formulate these models. Here in Chapter 6, I investigate the effects of mechanical disturbance on different fish taxa, both on the GBR and at Phuket, in order to identify effects that are generally applicable.

## 1.4 THE RESTORATION OF CORAL HABITATS

The practice and theory of actively restoring damaged habitats is becoming a major topic of interest to ecologists and managers (Jordan, 1995). In a practical sense, the definition of restoration ecology is, "returning the ecosystem to the condition before damage took place" (Cairns, 1995). To many, restoration is seen as a critical test of ecological theory, as it requires a fundamental understanding of the inter-relationships among organisms and between organisms and their habitat (Simberloff, 1988). As in basic ecological research, in developing principles of restoration, experiments must address basic hypotheses about key species and processes in the establishment and maintenance of community structure (Jordan 1995). Knowledge of the importance of habitat structure is critical to developing methods of restoration, and the lack of this knowledge may explain the absence of many completely successful attempts at restoration (Bohnsack, 1991). In most cases, restoration activities focus solely on the active rehabilitation of habitat-forming organisms (e.g. mangroves, corals, saltmarshes), and the effects on other components of the system are poorly understood (but see Clark & Edwards, 1994).

Coral reef ecosystems may have the ability to recover from degradation, but even in the absence of any further damage, natural recovery may take decades (Done, 1987, 1992). Active restoration could accelerate this process. In response to severe and multiple impacts, irreversible changes to coral habitats may occur (Hughes *et al.*, 1987; Kuhlmann, 1988; Hughes, 1994). In these cases, habitats may not recover from human disturbance without active manipulation (Pratt, 1994), but this has not been investigated. While there has been a small number of attempts to restore coral reef habitat on a limited scale, using coral transplantation (Yap & Gomez., 1984, 1985; Harriot & Fisk, 1988a, 1988b; Yap *et al.*,1992), there have been less attempts to consider the effects of habitat restoration on fish assemblages (Clark & Edwards, 1994). Is habitat restoration sufficient, or is the active rehabilitation of fish communities necessary? In Chapter 7, I consider the influence of active habitat rehabilitation on different fish taxa, both on the GBR and at Phuket, in order to identify effects that are generally applicable.

## 1.5 FISH-HABITAT INTERACTIONS AND THE "HEALTH" OF CORAL REEFS

Clearly, the linkage between habitat structure and fish communities, the effects of disturbance on communities and the methods of restoring communities represent vital topics in the management of coral reefs. However, detailed research programs on these topics will inevitably concentrate on only a few areas. The coral reefs around the world, especially in developing countries are threatened by various sources of anthropogenic disturbance (e.g. Tomascik & Sander, 1987; McClanahan & Muthiga, 1988; Guzman & Jimenez, 1992; Brown *et al.*, 1993; Wilkinson *et al.*, 1993; Hunter & Evan, 1995). These disturbances, however, are likely to vary across all spatial scales (local-regional-global), and thus must be integrated into regional management plans. This highlights the necessity to provide broad-scale information on the status or health of coral reefs, which may be indicated by the types of assemblages present.

Benthic assemblages appear to be the most basic component of coral reef systems, as they are the dominant bio-physical feature (Smith & Buddemeier, 1992) and provide habitat structure as well as primary productivity to the system. Estimates of living hard coral cover are usually used as the sole indicator of reef status (Brown & Howard, 1985; Wilkinson *et al.*, 1993), although sometimes, single fish taxa (e.g. butterflyfish) are used (Reese, 1981). There are a number of potential problems with this approach. Coral cover along may poorly represent what is happening to the whole community and may mask considerable underlying variability. Also, it does not take into account the different types of reef development, where many areas may be unsuitable for coral growth. Clearly, the status of coral reefs can only be measured by more detailed information on community structure and relationships among different components of the reef system. At the same time, complex information must be reduced to a small number of parameters or indices in order to provide managers with a clear way of ascribing conservation or management importance to different areas. In Chapter 8, I develop a method for evaluating the status of corals reefs using multiple information level approach and using benthic reef assemblage as base model.

## 1.6 MAJOR OBJECTIVES

The rationale for this thesis was to investigate the nature and strength of interactions between coral reef fishes and their benthic habitat, and examine the potential application of this knowledge in coral reef management. In an attempt to identify general patterns in different regions with a different regional pool of species, similar studies were carried out on the central GBR (Australia) and at Phuket (Thailand). To do so, it was necessary to establish observational procedures for quantifying fish-habitat relationships, and conduct common experiments on the effects of disturbance and habitat restoration in the different regions. The specific objectives for each data chapter were to:

- Develop an optimized sampling scheme for the quantitative description of fish assemblages, habitat structure and describing fish-habitat relationships (Chapter 3).
- Investigate fish-habitat relationships across multiple spatial scales and among years at Phuket, Thailand (Chapter 4).
- Investigate fish-habitat relationships across multiple spatial and temporal scales on the fringing near-shore reefs of the central Great Barrier Reef, Australia (Chapter 5).
- Simulate the effects of small-scale disturbance to habitat by reducing the cover of different components of the habitat and investigate the impact on the structure of fish assemblages. General patterns were sought by repeating experiments at two geographic regions (Phuket, Thailand and Orpheus Is., central GBR) and two localities within each region (Chapter 6).
- Simulate the effects of small-scale restoration of habitat by increasing the cover of different components of the habitat and investigate the impact on the structure of fish assemblages. Again, general patterns were sought by repeating experiments at two geographic regions (Phuket, Thailand and Orpheus Is., central GBR) and two localities within each region (Chapter 7).

- Develop a management decision tool for coral reef base on ecological data (Chapter 8).

# CHAPTER 2

# STUDY AREA

## 2.1 GENERAL BACKGROUND

### 2.1.1 Thai coral reefs

Coral reefs in Thailand are similar to those in other ASEAN, which are significant from both ecological and socio-economic viewpoints. It has been estimated that 30% of the world's coral reefs are found in the ASEAN region (McManus, 1988), that this regions harbours the greatest diversity of coral reef organisms (Longhurst & Pauly, 1987) and that these reefs have been rapidly degraded over the past 10 years (Wilkinson *et al.*, 1993). Coral reefs in Thailand can be divided into two general types: reef-building and non-reef coral communities. Most reefs are fringing reefs, with the greatest development on the west coast of the Thai Peninsula, in the Andaman Sea (Ditlev, 1978; Chansang *et al.*, 1985). Reefs in this area are in a nutrient poor environment due to lack of major river systems and direct exposure to the Indian Ocean. Non-reef coral communities are described as areas where coral grow on hard substratum with no substantial limestone reef development. Non-reef coral communities can be found elsewhere within the coastal area in the Gulf of Thailand (Sakai *et al.*, 1986). This area is greatly influenced by freshwater run-off from four major rivers. Thus, these corals develop in a nutrient rich environment.

The socio-economic significance of coral resources in Thailand lies primarily in the fisheries associated with reefs, especially small scale, fishing activities. Although from a purely economical perspective, the value from this activity may not be high (see Spurgeon, 1992), it is very important for local communities especially in terms of social values and subsistence. The economical value of the reef is more obvious from the point of view of tourism, which has developed very rapidly during the last two decades (Sudara & Yeemin, 1994), and also sea shell trading (Sudara & Nateekarnchanalap, 1988).

The growing population and economy of Thailand has contributed to widespread destruction of natural resources, including coral resources. There is a strong potential of human impact especially on near-shore coral reefs/assemblages. The main impacts, their relative importance varying among localities, include illegal fishing, tourism (Garces, 1992), tin mining (Chansang *et al.*, 1981, 1985), and channel dredging (Brown *et al.*, 1990). Other activities that may not destroy coral immediately, but have long term effects, include the effect of sewage input and nutrient enrichment, and the collecting aquarium animals using poisons. The continuing degradation of this ecosystem reflects the ineffectiveness of existing management plans, due to lack of ecological information, the absence of enforcement and socio-economic (Garces, 1992). Degradation of coral reef resources in Thailand is coming from both lethal and chronic impacts, which makes self recovery difficult. Thus, solely protective management strategies to conserve these resources may not enough and the restoration or active recovery of these resources may be necessary. It has become clear that a management strategy should be specifically developed for Thai conditions, both in term of resources and socio-economic structure of the country (and also local area).

The study on coral reefs in Thailand started in the mid of 1970's. Earlier works focused on taxonomy of reef organisms e.g. corals (Ditlev, 1976), algae (Egerod, 1974, 1975), Molluscs (Neilsen, 1976) and fishes (Mongkolprasit & Sonthirat, 1980; Mongkolprasit, 1981). Studies on the morphology of coral reefs was only conducted after some taxonomic information had accrued (Sudara, 1977; Ditlev, 1978). After 1980's, quantitative reef surveys were conducted at many locations, but most of them were limited to benthic coral assemblages (Chansang *et al.*, 1981, 1985; Srithunya *et al.*, 1981; Sudara, 1981; Brown & Holley, 1984). The other assemblages that received attention during the late 1980's included coral reef fishes (Menasveta *et al.*, 1986; Nakasone & Manthachitra, 1986; Manthachitra, 1991; Manthachitra & Sudara, 1991; Satapoomin, 1993), crustaceans (Nakasone *et al.*, 1986) and algae (Kamura & Choonhabandit, 1986). Ecological information was also collected during this period to tackle potential damage from human activities, e.g. the impacts from tin mining activities (Chansang *et al.*, 1981; Brown & Holly, 1982), port construction and channel dredging (Brown *et al.*, 1990, 1993; Clarke *et al.*, 1993). Biological information associated with these impacts is relatively scarce (Charuchinda & Hylleberg, 1984; Brown *et al.*, 1994). Most research efforts have concentrated on applied issues (e.g. reef monitoring) rather than basic ecological questions due to the urgency of the problem and limited funding, a continuing problem for developing countries.

The reef monitoring program was first established first in the Andaman Sea in the early 1980's (Chansang & Phongsuwan, 1993). In 1985, the first long-term and broad scale monitoring of the status of coral reefs in Thailand was initiated, under the ASEAN-Australia Cooperative Project on Coastal Living Resources, with the plan to cover most of the reef area (English *et al.*, 1988). This project aimed to provide an extensive baseline database of the major living resources with emphasis on management approaches. This project not only provided the first long term monitoring information on coral reef conditions, but also provided a new standardized methodology for coral reef surveys in Thailand and the ASEAN region (Dartnall & Jones, 1986). The results from this project have also been used to create a national policy for the conservation of coral reef ecosystems in Thailand, under the ASEAN-US Coastal Resources Management Project (Lemay *et al.*, 1991; Lemay & Hale, 1993). The prominent success of both projects was highlighted in 1991 when the Thai cabinet passed the national policy for coral resources conservation as an urgent issue.

Because the previous monitoring program involved several research institutions (CREST, 1989; CRT, 1989; Chansang & Phongsuwan, 1993; Geater *et al.*, 1994), there were several problems involving the application of the data for both scientific and management purposes. These problems were: 1) different sampling schemes, 2) variable data quality due to different sampling techniques and designs, and 3) inconsistent *criteria* for management decision making.

Under the national policy, most of the future efforts will be focus on both protection and restoration of coral resources, and also include initiation of public awareness. However, any future plans will require solid scientific knowledge especially in term of ecological and biological information. For example, the protection of coral reefs requires a standardised monitoring program and requires an effective "indicator" approach to rapidly assess the status of this ecosystem. The restoration program needs a cost effective technique for the rehabilitation of reef ecosystem under various conditions. However, the existing information is inadequate. Because of the socio-economic importance of the resource, the protection and restoration of coral reefs in this region are urgently required.

### 2.1.2 Australian coral reefs

In contrast to Thailand, the coral reef systems on the Great Barrier Reef have been intensively studied and management practices are well established and effective (Dinesen, 1988; Lassig *et al.*, 1988; Kelleher, 1993). The Great Barrier Reef (GBR) is one of the biggest reef systems in the world. It covers almost 349,000 km$^2$, stretches over 2,000 km and includes more than 2,900 individual reefs (Kelleher, 1993). The taxonomy of reef-building and associated organisms have been well (e.g. Veron & Pichon, 1976, 1979, 1982; Veron & Wallace 1984; Randall *et al.*, 1990). The reef morphology has been described in detail, in terms of geology (Hopley, 1982) and coral assemblages (Done, 1982). The cross-shelf patterns of other assemblages have also be described, e.g. fish (Williams, 1982; Williams & Hatcher, 1983; Russ, 1984a, 1984b), sponge (Wilkinson & Cheshire, 1989), and crustaceans (Preston & Doherty, 1994). A large body of basic and applied ecological information has accumulated.

Management practices were first established in 1903, under the State Forest and National Parks Act. However, an integrated regional approach was initiated in 1975, with the establishment of the Great Barrier Reef Marine Park Authority (GBRMPA, 1981). Management implementation involves various activities especially monitoring (Lassig, *et al.*, 1988) as well as day-to-day management (Dinesen, 1988).

The threats to the GBR are comparatively small when compared with Thailand, primarily because of the difference in the size of the human population in proximity to the reefs. The major factors that modify habitat structure on the GBR appear to be natural processes, particularly outbreaks of the crown of thorn starfish (Kenchington, 1976: Moran, 1986) and cyclones. The effects from human activities are local and less potentially damaging to habitat structure, including anchor damage, tourism, over-fishing and research (Dinesen, 1988). Potential threats of terrestrial agricultural practices and run-off from the land, which may lead to eutrophication on inshore reefs, is being investigated (Bell, 1991; Steven & Larkum, 1993). In general, the relative lack of problems from human impact can be attributed to lower population densities and good management systems, e.g. reef zoning, monitoring and law enforcement. Another important factor is the socio-economic background, which can be characterized by a relatively high public awareness of environmental issues. Additionally, there is research to support both management decisions e.g. routine reef monitoring (Moran, *et al.*, 1991) and the study of future threats, e.g. crown of thorn starfishes (Zann & Moran, 1988) and eutrophication (Steven & Larkum, 1993).

## 2.2 STUDY AREA

### 2.2.1 Phuket, Thailand

Phuket is Thailand's largest island in the Andaman Sea (8□ N, 98□ 20□ E) and is separated from the mainland by a channel of only 200 m width. The climate of Phuket is tropical, with a strong monsoonal influence. The northeast monsoon brings the cool dry season during November to March and the southwest monsoon brings the wet season from May to October. Surface sea temperature is about 25 - 29□ C, salinity is 29 - 33 ppt and tide is semidiurnal with an average spring range approximately 2.4-

2.7 m (Phongsuwan & Chansang, 1993). Coral reefs are located mainly on the west and south coasts, and around near-shore islands. Reef physiomorphology differs due to the influence of monsoon. Reefs in the south and on the east coast are protected and develop under shallow turbid waters of 4 - 5 m depth with an extensive intertidal reef flat (Ditlev, 1978; Brown *et al.*, 1990). In contrast, reefs on the west coast are exposed directly to the southwest monsoon from Indian Ocean. Reefs have developed under clear water with extensive reef slopes extending to 15 m deep (Chansang *et al.*, 1981; Chansang & Phongsuwan, 1992). Human impacts were extensive and intense, stemming from various activities, such as tin mining (Chansang *et al.*, 1981; Chansang, 1988), tin smelting (Brown & Holley, 1982), port construction and dredging (Brown *et al.*, 1990). Management measures to conserve coral reefs exist but are largely ineffective due to limited enforcement.

The description of spatial and temporal pattern of fish assemblages and habitat structure (Chapter 4), as well as the experiments on the effect of habitat degradation (Chapter 6) and rehabilitation (Chapter 7) were carried out at Phuket during 1994 to 1995 (Table 2.1). The spatial and temporal studies were conducted along the west and south coasts of Phuket. A total of eight study localities, four each along the west and south coasts of Phuket Island, were sampled (Fig. 2.1B). They are Niyang (A), Kamala (B), Patong (C), Kata (D), Lon Island (E), Hi Island (F), Aoe Island (G) and Maiton Island (H). Two habitats, middle reef slope and reef edge, were studied at each locality. These two habitats can be defined by water depth, with the reef edge at 3 m below mean sea level (MSL) and the middle reef slope at 5 - 6 m below MSL. Data were collected twice, March 1994 and March 1995, aiming to detect annual variation.

An experimental manipulation of habitat was carried out at Maiton Island (7□ 45□ N, 98□ 29□ E), 8 km southeast of Phuket. Two localities on the east of the island were selected and labeled as "Northeast reef" and "Southeast reef" (Fig. 2.1C). These sites were chosen for because the reefs are heavily degraded and are presently protected by a resource operator. Experiments were set up on the reef slopes at approximately 4 - 6 m below MSL.

### 2.2.2 The Great Barrier Reef, Australia

This study focused on the reefs fringing inshore islands in the upper central section of the Great Barrier Reef (18□ S to 19□ 10□ S, 146□ E to 147□ E). The islands, all within 20km of the coast, included Orpheus (16 km), and Magnetic (5 km).

The influence of human activities on reefs of this region is likely to come primarily from run-off from land. At Orpheus Island, the potential source of impact might be related to agricultural activities (sugar cane plantation), while at Magnetic Island, it might be largely due to domestic water run-off and coastal construction activities. The direct impact from recreational fishing, tourism and boat anchoring is likely to be slight and localized.

The studies carried out on the Great Barrier Reef are summarized in Table 2.1. The optimization of transect techniques was carried out at Orpheus Island (Chapter **3**). Three habitats (reef flat, reef edge, and reef slope) were defined at Cattle Bay and Hazard Bay during October 1992. Studies of spatial patterns in benthic reef and fish assemblages (Chapter 5) were conducted at 13 localities from November 1993 to January 1994. Nine localities were studied at Orpheus Island and four at Magnetic Island, in proportion to the amount of reef development (Fig. 2.2). Two habitats (middle reef slope and reef edge) were surveyed for benthic reef and fish assemblages at each locality. The middle reef slope was usually at a depth of 6 - 8 m below mean sea level (MSL) whereas the reef edge was 3 - 4 m below MSL.

Studies of temporal pattern were conducted on the reef slope and reef edge at three localities of Orpheus Island; Cattle Bay, North Pioneer Bay and South Pioneer Bay. Data was collected at two to three monthly intervals for a total of 10 times from December 1993 to December 1995.

Experiments to investigate the influence of habitat degradation (Chapter 6) and rehabilitation (Chapter 7) on reef fish assemblage were set up on the reef edge of Cattle Bay and Pioneer Bay where the depth is approximately 3 - 4 m below MSL. These sites were chosen because they had relatively poor status compared with other localities in Orpheus Island, and were protected from strong wind and wave action. The study was conducted during October 1993 to December 1995.

Hereafter, the term "study site" will be used for each habitat at each locality. All of the study localities can be considered as near-shore environments which usually have a high level of influence from terrestrial conditions and human activities.

Table 2.1. Summary of the studies performed in this thesis.

| | Thailand | | Australia | |
|---|---|---|---|---|
| Study/Area | Phuket | Maiton | Orpheus | Magnetic |
| Optimizing (Chapter 3) | | | Aug'92 | |
| Spatial (Chapter 4-5) | Mar'94-95 | Mar'94-95 | Dec'93 | Jan'94 |
| Temporal (Chapter 4-5) | Mar'94-95 | Mar'94-95 | Dec'93-95 | |
| Experiment (Chapter 6-7) | | Mar'94-95 | Oct'94-Dec' 95 | |
| Indices (Chapter 8)* | Mar'94 | Mar'94 | Dec'93 | Jan'94 |

* data from the inner Gulf of Thailand (Manthachitra, 1994) were also used.

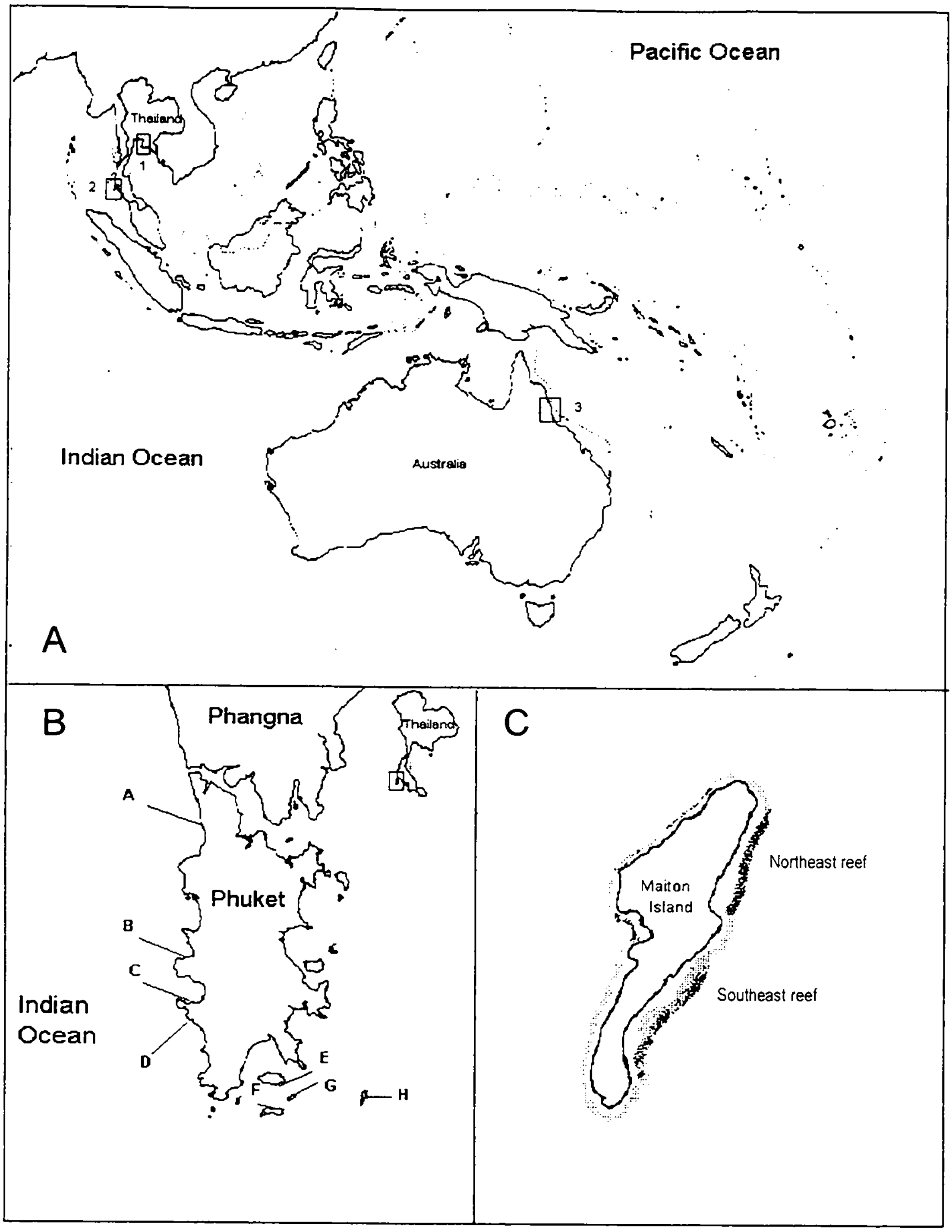


Fig. 2.1. Maps showing study areas: A) Southeast Asia and Australia, B) Phuket, Thailand, and C) Maiton Island (Phuket - H).

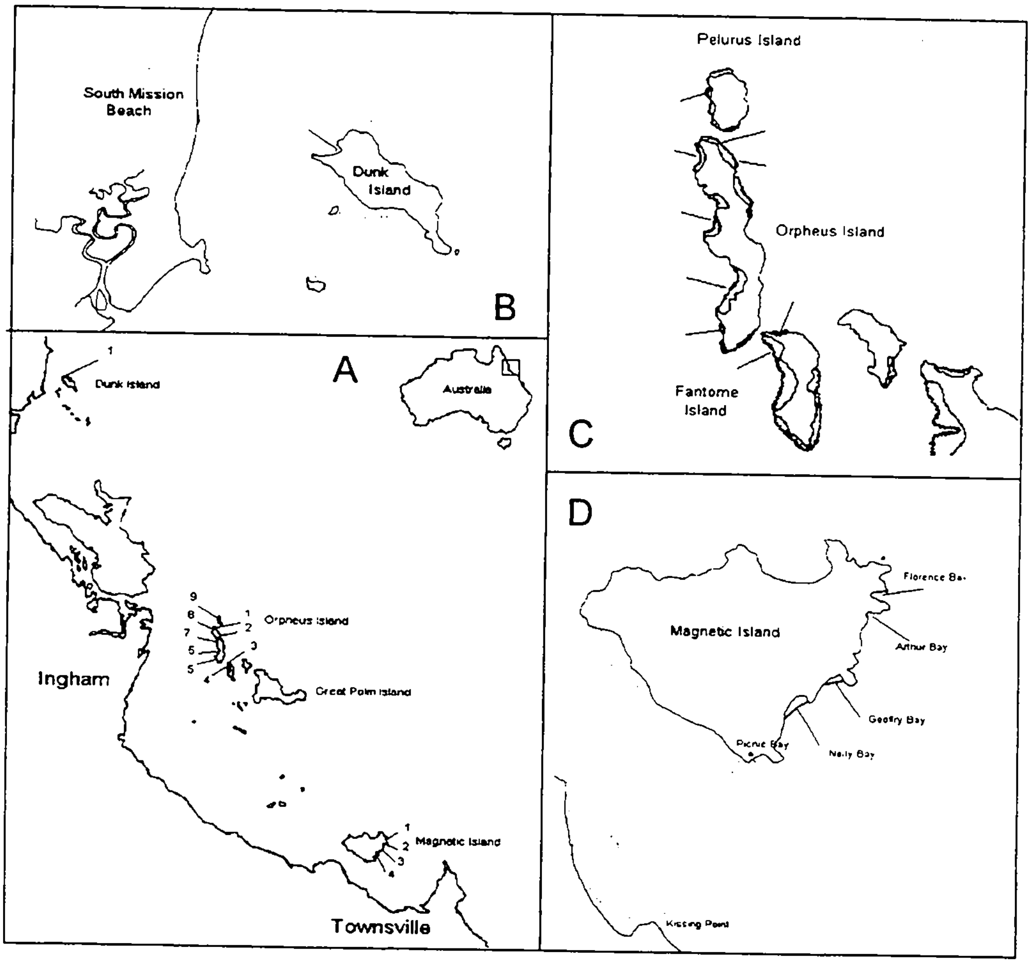


Fig. 2.2. Maps showing study areas at the central Great Barrier Reef, Australia: A) the central Great Barrier Reef, B) Dunk Island, C) Palm Islands, and D) Magnetic Island.

# CHAPTER 3

# OPTIMISING TRANSECT SAMPLING FOR QUANTIFYING FISH-HABITAT RELATIONSHIPS ON CORAL REEFS

## 3.1 SUMMARY

Different transect methods and dimensions for quantifying fish and benthic assemblages on coral reef were independently optimised. Data were then used to determine the combination of methods providing the most consistent patterns in quantifying fish-habitat relationships. The optimisation trials focused on the accuracy, precision and efficiency of sample estimates (i.e. the most cost-efficient combination of sample unit size, method and number for achieving the desired precision). Trials were conducted to examine: (a) the effects of transect length on estimates of fish abundance and coral cover, (b) the effects of transect width on estimates of fish abundance only, and (c) the effects of 2 different transect methods on the estimates of benthic biotic cover only. Trials were conducted at three reef habitats (reef flat, reef edge and reef slope) at two localities (Cattle Bay and Hazard Bay, Orpheus Island) in order to assess variation in the optimum sample unit size and number. Estimates of abundance were calculated for four species of wrasse; *Halichoeres melanurus, Halichoeres chloropterus, Thalassoma lunare* and *Stethojulis strigiventer*. The best combination of eight transect dimensions, based on two transect widths (2 m and 5 m) and four transect lengths (30, 50, 70 and 100 m) was assessed. The influence of transect width varied among localities and habitats. The survey using a 2 m wide transect estimated higher abundance than that of 5 m width on the reef edge and reef slope of Cattle Bay, but provided less precise and lower efficiency than the 5 m transect. In contrast, transect length did not show a significant effect on the estimates of the mean. Short transects were more likely to provide less precise estimates, but were more efficient than a longer transects.

Estimates of benthic cover were assessed using both Fixed Density Point Transects (FDP), in which lifeforms under points along transect are recorded, and Line Intercept Transect (LIT), in which lifeforms under actual distance along transect line are recorded, were compared . Benthic cover was divided into eight major benthic categories: *Acropora* coral, Massive coral, Branching coral, Minor living component, Fire coral, Soft coral, Dead coral, and Dead coral with algae. The results showed that FDP was more

efficient than LIT (i.e. same precision can be achieved for lower cost). However, the more precise estimates of FDP were discounted by an apparent inaccuracy in its estimation of less common categories, which could be missed. Selection of the appropriate technique is, therefore, dependent upon the objective of the study, with FDP preferable when sampling common species and LIT better for sampling rare species. A 30 m transect was the most efficient length for both sampling techniques.

Optimisation sampling for multiple species/categories was primarily based on, 1) a qualitative approach for selecting an appropriate transect dimension, 2) a quantitative approach for providing the optimal number of replicates across multi-spatial scales, and 3) a compromise for obtaining the optimal number of replicates across multiple species. On this basis, the most efficient method of sampling the four wrasse species was 5 x 30 $m^2$ which required eight replicates for a precision level of 0.2. The 30 m transect was also the most efficient for benthos which six and five replicates are required for FDP and LIT respectively.

The influence of sampling schemes (techniques and transect dimensions) on the quantitative description of fish-habitat relationship using canonical correlation analysis showed that only high-precision sampling schemes provided consistent results. The preferred sampling scheme to investigate the fish-habitat relationship, therefore, relies on seven replicates of a 30 m transect when the corresponding fish survey is conducted on a 5 m wide transect. The habitat survey can be conducted using either FDP and LIT.

## 3.2 INTRODUCTION

Visual fish counts have been useful in identifying common spatial patterns in the structure of coral reef fish assemblages, such as variation among reef localities and among reef habitats (Goldman & Talbot, 1976; Williams, 1982; Russ, 1984a, 1984b; McCormick & Choat, 1987; Roberts *et al.*, 1992; McClanahan, 1994; Meekan *et al.*, 1995). However, the same approaches have provided a far less clear picture of the relationships between fish abundance and particular habitat variables. Some studies have found strong correlation (positive or negative) between fish abundance and measures of habitat structure (Luckhurst & Luckhurst, 1978; Carpenter *et al.*, 1981; Bell & Galzin, 1984; Bouchon-Navaro *et al.*, 1985; Roberts & Ormond, 1987; McCormick, 1994) whilst others have found little or no correlation (Luckhurst & Luckhurst, 1978; Bell *et al.*, 1985). The most likely reasons for this inconsistency are differences in methodologies and the reliability of the data. Some studies have described the relationship between fishes and habitat by using the quantitative data on the fish, but only semi-quantitative data on the habitat (Kaufman, 1983; Walsh, 1983; Sano *et al.*, 1984, 1987; Williams, 1986; Dawson-Shepherd *et al.*, 1992). The differences in the quantitative relationships between fish and habitat may due to the sampling techniques and designs (McCormick, 1994). Investigation of this relationship must be based on accurate and precise data on both fish and habitat components. However, procedures for optimising sampling designs to get the most reliable estimates for the minimum cost, are carried out at most on the fish component. Appropriate methods for the best method for detecting relationships have not been considered at all.

Quantitative studies of coral reef fish assemblages are complicated by their mobility and diversity (Sanderson & Solonsky, 1986). Visual estimates, however, have been adopted as one of the most efficient and non-damaging tools for studying these fish assemblages (Brock, 1954). The primary assumption of this method is that "you do not count what you do not see" (Russell *et al.*, 1978). This assumption leads to the conclusion by some that the visual census method under-estimates the true value. Thus, the most accurate estimates is usually considered to be the highest value (Sale & Sharp, 1983). A number of different visual census methods have been widely used for coral reef fish studies, including strip-transects (Bell & Galzin, 1984; Roberts & Ormond, 1987; Fowler, 1987), timed fish counts (Williams, 1982; Russ, 1984a) and species-time censuses (DeMartini & Roberts, 1982). Amongst these techniques, line transects appear to be the most widely used but their dimensions (width & length) have varied greatly, usually for reasons not disclosed by authors.

A wide range of factors that may influence the estimates of abundance gained from a line transect have been investigated (Fowler, 1987; St. John *et al.*, 1990). These factors include transect dimension and number of replicates, behavioural characteristics of each target species, the sensitivity of the observer, the number of species and their relative abundance, and the topography of the areas being sampled. However, transect dimension is likely to be the most important and controllable factor. Sale & Sharp (1983) found a negative association between the density of fish and width of transects for five unrelated species. Fowler (1987), in contrast, found that transect width and length did not significantly influence the density estimates of chaetodontids. This indicated that the transect width is influenced by the conspicuousness of morphology or behaviour of the focal species (Sale & Sharp, 1983; Fowler, 1987; McCormick & Choat, 1987). No single transect dimension, therefore, is suitable for all species and maturity stages of coral reef fish or all habitats. Some compromises have to be made.

Increasing the number of replicates increases both the precision and cost of an abundance estimate (Pringle, 1984; Downing & Anderson, 1985; Bros & Cowell, 1987; Fowler, 1987; McCormick & Choat, 1987), and may increase the variance explained in quantitative description of fish-habitat relationships. Therefore, it is necessary to determine the optimum sampling scheme (combination of transect dimension and number of replicates that achieves the desired precision for the minimum cost - see Pringle, 1984; Andrew & Mapstone, 1987), especially when quantitative information on fish-habitat relationships is needed. However, all of the above concepts have been applied to only single species/variable situations, even when the study essentially involves a multiple species assemblage (McCormick, 1995). It is not feasible to use different transect dimensions to sample different species in such communities (Mundy, 1991). There have been few attempts to optimise effort in whole community studies, except for univariate measures of community structure, such as species richness and biomass (Peterson & Rabeni, 1995).

Biotic habitat structure is usually measured as the surface coverage of the dominant organisms occupying the substratum (Loya, 1972). Among a number of methods used in the quantitative study of coral reefs, the line transect technique is the most widely used (Loya, 1972, 1976; Bouchon, 1981; Benayahu & Loya, 1981; Bradbury *et al.*, 1986; Mapstone *et al.*, 1989; Brown & Suharsono, 1990; Mundy, 1991; English *et al.*, 1994). This technique was developed in the field of terrestrial plant ecology (McIntyre, 1953) and has been adopted by coral reef ecologists, where it is usually referred to as the "line intercept transect (LIT)" method (Loya, 1972, 1978; Marsh *et al.*, 1984). The underlying assumptions of this technique are that the size of the object is small relative to the length of the line, and that the length of the line is small relative to the area of interest (Mundy, 1991). The benthic assemblage can be described by using both taxonomic groups and life-form categories (which provide a morphological description) depending on the objective of the study. The percentage cover of each particular substratum can be determined by calculating the fraction of the length of the line that is intercepted by the substratum.

Line intercept transects can be very time consuming. The original technique has also been modified by many workers (e.g. Bainbridge & Reichelt, 1988) with the aims of decreasing sampling time or increasing efficiency. One modified method is called "Random Point Transect (RPT)" method as it uses a number of random points along the transect line and records the frequency of substratum instead of actual fractional length (Bainbridge & Reichelt, 1988; Kaly & Jones, 1994). However, the relative performance of point transects versus line intercept transects in terms of accuracy, precision and efficiency has never been studied. The appropriate length of transects, the number of points used and the number of replicates required, all need to be determined.

The problems with previous work in which transect techniques have been used are: 1) no or insufficient replication, 2) use of transects that are too short, and 3) insufficient awareness of spatial variation leading to extrapolations about general patterns from one or two sites (Mundy, 1991). Certainly these problems, which are related to each other, can be solved by following an optimisation procedure (e.g. Pringle, 1984; Andrew & Mapstone, 1987). Mundy (1991) provided the information on the precision and efficiency of several transect lengths on the estimation of benthic area cover and applied them to optimise the sampling scheme. However, although Mundy's (1991) study was conducted across several sites (reefs), it was restricted to one habitat and conclusions were based on qualitative and univariate quantitative approaches.

The aim of this study was to determine the optimum sampling scheme (technique, transect dimensions, and replicate number) for estimating the abundance of fish and cover of benthic organisms, for examining fish-habitat interactions. The best sampling design is assessed in terms of accuracy, precision and efficiency. The influence of different fish and habitat survey techniques on the stability of descriptions of fish-habitat relationships was also assessed using multivariate techniques.

## 3.3 MATERIALS AND METHODS

### 3.3.1 STUDY AREA

This study was carried out at Orpheus Island (18° 35′ S, 146° 29′ E), the central section of the Great Barrier Reef (Fig. 3.1A). Two localities were studied, Cattle Bay and Hazard Bay. Each locality was stratified into three habitats i.e., reef flat, reef edge and reef slope. The classification of these habitats is primarily based on the similarity of benthic substrata and reef depth (Fig. 3.1B). The reef flat is the area that has a depth of around 0 - 0.5 m above chart datum and is characterised by dead coral. The depth of the reef edge is around 0 - 0.5 m below chart datum and this area is mostly covered by soft coral and a number species of hard coral. The reef slope lies at a depth of around 3 - 4 m which is also dominated by soft and hard corals.

### 3.3.2 TARGET SPECIES /GROUP

Based on a preliminary survey, four species of small wrasse from three genera were selected for the study based on their predictable occurrence (Table 3.1). Benthic substrata were categorised into 10 major categories, based on the growth forms described by Veron (1986) (Table 3.1).

### 3.3.3 SAMPLING TECHNIQUE AND DESIGN

For fish counts, fiberglass measuring tapes were laid out to appropriate lengths in a direction parallel to the shore line at each sampling site (habitat) at a fixed depth. Fish were counted using the simultaneous technique (Fowler, 1987), in which fish are counted at the same time as the tape is unwound. The sampling design used was a balanced multi-stage design (factorial) in which there were four factors; locality, habitat, transect width and transect length, taken into account to determine the influence of each factor on the abundance estimates of the small wrasse species. Two transect widths, 2 m and 5 m, were examined. These transect width sizes were considered for three main reasons. Firstly, the size of fishes was relatively small, especially at their initial phase and therefore the area had to be able to be search carefully. Secondly, the water visibility at the study area was low due to their near-shore position. Lastly, reef physiomorphology shows narrow distinctive zones (Fig. 3.1B) which wide transect may overlap with other habitats. Four arbitrary transect lengths were used: 30, 50, 70, and 100 m. Four replicates of each of the eight different transect dimensions were carried out within each habitat and location.

For the benthic assemblage, the same four different transect lengths and two sampling techniques, the Fixed Density Point Transect (FDP) and the Line Intercept Transect (LIT) were compared. As with fish counts, sampling was repeated in each of the three habitats at each location, with four replicates, giving a balanced multi-stage design. Both transect techniques were performed on the same line transect that was used concurrently to census the fish.

The FDP transects were conducted by using a density of one point per meter. However, the point was not fixed, but could move at random within a 1 m segment of the transect line (stratified random sampling). This floating point technique ensured random sampling along the full distance of the transect. Each point was randomly located within one meter segments of the transect line. The benthic categories falling under each random point were recorded. The cover of each category was calculated as the proportion of the total number of points falling on each habitat category. Using the LIT method, the benthic categories falling under the transect were recorded as actual distance measurements along the transect line. The total distance of each category was added up and expressed as a percentage of the total transect length.

The time taken to complete each transect (sampling time) was recorded, for both fish and benthic surveys. However, the overall mean sampling times (T) were calculated across all localities, habitats and replicates of each transect dimension. Sampling time is defined as the time used to collect data and does not include time spent retrieving the measuring tape because the complication of conducting several sampling techniques at once. Mean sampling time was used to evaluate the cost-efficiency of each sampling scheme performed in this study.

### 3.3.4 PRECISION AND EFFICIENCY DETERMINATION

The ability and precision of all estimates were assessed by examining Standard Error (SE) and Precision Value (PV = SE/mean) of the untransformed data. These were calculated from the four replicates of standardised abundance data for each combination of all factors.

The cost-efficiency of each transect dimension was assessed by examining total sampling time (optimal number of replicates x mean sampling times) required to achieve at target precision level. The optimal number of replicates was calculated as $n = [\ S / (\bar{X} * PV)]^2$ where, n = optimal number of replicates, S = sample standard deviation, $\bar{X}$ = sample mean, and PV = required precision level (Andrew & Mapstone, 1987).

The optimal number of replicates at the required precision level was rounded up to the next whole number (e.g. 3.1 = 4) and then used to calculate total sampling time. The transect dimension that required the shortest time to complete the sampling procedure at the required level of precision is considered to be the most efficient.

As this study involved a number of species and occurred over multiple spatial scales, each species needed to be compared to determine the best overall sampling scheme. There were two steps used for making this decision: Firstly, the "mode" was used to select the appropriate transect dimension. That is, I chose the transect dimension that came out as being the most efficient with the highest frequency across all species, habitats and localities. Secondly, the "average" was used to decide on the number of replicates. The optimal number of replicates of the preferred transect dimension for each species was averaged across all localities and habitats. Transect dimensions which failed to detect any fish (PV has no value) are treated in the same way as dimensions which detected fish in low numbers and frequencies (rare species). When PV has no value, the number of replicate cannot be calculated and cannot be presented with zero. If they were omitted from the calculation, number of replicate would higher than it should be. If zero PV were used in this case, the average number of replicate would lower than it should be. If enough replicates were employed, however, rare species could be detected. But the precision of the estimate would considerably low. In this case, PV = 1 was used to represent undetected sampling because it means the size of error equal mean which can be considered as the lowest data quality. At PV = 1, therefore, a maximum number of replicates can be calculated for an undetected sampling. Maximum number of replicates can be calculated by the following formula derivation:

when $PV = 1$:

$$SE = X = S/\sqrt{n_l}$$

then

$$S = SE*\sqrt{n_l} = \bar{X}*\sqrt{n_l}$$

thus

$$n_m = [S/(\bar{X}*PV)]^2 = [(\bar{X}*\sqrt{n_l}) / (\bar{X}*PV)]^2$$
$$= [\sqrt{n_l} / PV]^2$$

where

$n_l$ = number of replicates used in the pilot study

$n_m$ = maximum number of replicates at required precision level

A compromise among species was necessary to arrive at a common number of replicates to survey all fish species or categories of benthic organisms. The number of replicates was weighted on the basis of the relative total number of each species in the data matrix. The most abundant species was given a weighting equal to one while other species were weighted relative to the most abundant species. The optimal number of replicates for each species were thus adjusted before being averaged. This was done by multiplying each number of replicates by the species "weighting factor". This means optimisation of the sampling scheme was done with respect to the abundance of the most common species. The main objective of this method was to calculate the required numbers of replicates of the selected transect dimension for a given level of precision. A workable level of precision was selected base on the feasibility of the outcome replicates.

The weighting factor was applied because the basis of optimisation sampling scheme using precision value is for one variable (species). The application for community or multi-species studies need some modification. If the abundance of all species was taken into account, rare species will inflate the number of replicates which in most case impractical. If only the most common species was considered, logically, less common and rare species will completely be ignored from the process. By taking both problems into account and considering species composition in assemblage, weighting factor is an alternative way.

### 3.3.5 STATISTICAL ANALYSIS

The estimates of fish abundance and benthic cover were analysed using a Multivariate Analysis of Variance (MANOVA), which tested the four main factors: location, habitat, transect width and transect length. A balanced, fully orthogonal model was used in which all factors were treated as fixed factors. The Pillai's trace criterion was used to show the overall effect of each factor because it is more robust to heterogeneity of variance than comparable test statistics (Green, 1979). In all data sets, densities of fish were standardised as numbers per 500 $m^2$, for comparison. The abundance data were log (x+1) transformed to satisfy the assumptions of normality and homogeneity of variance (Underwood, 1981; Bray & Maxwell, 1982). The significance level used in this study was P = 0.05 but the lowest P was also reported. A MANOVA was carried out using Proc GLM in SAS (SAS Institute Inc., 1990). A Canonical Discriminant Analysis (CDA) was subsequently employed to help identify the nature of any significant differences found by MANOVA (Bray & Maxwell, 1982). A CDA was conducted based on a centered log (x+1) data matrix to remove the size effect. The canonical structure of each variable was used as a responsive factor for the discrimination of each group of all four factors in combination. Angular interpretation was then used to interpret the ordination plots produced by CDA. A CDA was conducted using Proc CANDISC in SAS (SAS Institute Inc., 1990).

The relationships between a multiple species of wrasse and benthic life-forms were determined by Canonical Correlation Analysis (CCA). The likelihood ratio analysis was carried out to determine the significant of relationships between two components. The redundancy analysis was also performed after CCA to indicate predictive/responsive roles between two components. The data set of both reef fish and habitat were log (x+1) transformed to fulfill the assumption of normality and to reduce the effect of high abundance species/groups. The relationships between fish and habitat were compared for fish and habitat data collected using the same transect length. Thus, the influence of the sampling technique on the relationship between fish and habitat components could be compared for 16 different combinations of fish and habitat data sets (Table 3.2). The results were considered in terms of both general relationships between the two components, as well as specific relationships between variables from each component. The calculation was performed using Proc CANCORR in SAS (SAS institute Inc., 1990).

## 3.4 RESULTS

### 3.4.1 FISH ASSEMBLAGE

#### 3.4.1.1 Abundance estimates

The results of the MANOVA indicated that the influence of transect width on wrasse abundance estimates is not consistent across habitat and locality (significant effect of Locality*Habitat*Transect Width interaction, Table 3.3). A CDA revealed the nature of this significant interaction (Fig. 3.2). The first two canonical discriminants, which explained 87.4% of total sample variation, showed that the separation distinction between transect widths occurred only at the reef edge and reef slope of Cattle Bay (i.e. CE and CS, Fig. 3.2). The main trends in species that were responsible these transect width effects were *H. melanurus, T. lunare* (Axis 1), *H. chloropterus* and *S. strigiventer* (Axis 2). At CE and CS, estimates using 2 m wide transects estimated more numbers of *H. melanurus, T. lunare, H. chloropterus* and *S. strigiventer* than estimates by 5 m wide transect (see also Appendix 3.1 - 3.4).

The MANOVA results indicated that there was no significant effect of transect length, or any interactions involving transect length (Table 3.3). The result of CDA showed trends in the influence of transect length that were not consistent across transect width, habitat and locality (Fig. 3.2). The separation visible at CF1 is caused by the abundance of *H. chloropterus*. The one at HS1 is explained by *H. melanurus* and *T. lunare*. The trends suggested that short transects give higher abundance estimates than long transects, although this was not statistically significant.

The distributional pattern of all four species of wrasse can also be seen from the results of CDA (Fig. 3.2). The separation between localities, Cattle Bay and Hazard Bay, may be due to *H. chloropterus* (Axis 2) which it is more abundant at Cattle Bay than at Hazard Bay. The separation between habitats can be attributed to *H. melanurus* and *T. lunare* which are more abundant at the reef edge than at the reef slope and reef flat respectively.

#### 3.4.1.2 Precision and efficiency

The most precise transect dimension for abundance estimates across habitat and locality varied among species (Table 3.4). For *H. chloropterus* and *T. lunare,* the 5 x 100 $m^2$ transect was the most precise transect dimension (4 from 6) especially in the estimates for the reef slope. The results for *H. melanurus* and *S. strigiventer* were not consistent across habitat and locality, but there were trends that suggested larger transect dimensions provide more precise estimates (low precision value) for the same number of replicates. For all species, the 5 x 100 $m^2$ transect was the most precise transect dimension (10 from 24). When considered transect width and length separately, the 5 m wide transect was more precise than the 2 m one (17 from 24) one, and the 100 m transect provided more precise estimates than shorter transects (12 from 24).

The mean sampling times for 30, 50, 70, and 100 m transects of fish visual census were 4, 7, 9, and 15 minutes respectively for a transect width of 2 m and 5, 8, 10, and 16 minutes for a transect width of 5 m.

The most efficient transect dimension for obtaining a precision value of 0.2 for abundance estimates across all habitats and localities varied between species (Table 3.5). A transect of dimension 5 x 30 $m^2$ was found to be the most efficient for *H. melanurus* and *T. lunare* but no conclusions could be made for *H. chloropterus* and *S. strigiventer*. For all species, inconsistent results arose more from transect length than from transect width. The major trends in transect length showed that shorter transects tend to be more efficient than longer transects, and 5 m wide transects were more efficient than 2 m wide transects (18 from 24).

From these results, it can be concluded that the 5 m wide transects were more suitable than the 2m transects for estimates of the abundance of the four wrasse species, in terms of both precision and efficiency. For transect length, a short transect (30 m) is preferred in terms of efficiency.

### 3.4.1.3 Optimal sampling scheme

Across all locations and habitats, *H.* ***melanurus*** accounted for 76.4% of the individuals counted, with the remainder taken up by *H. chloropterus* (7.6%) , *T. lunare* (12.1%), and *S. strigiventer* (3.9%). Estimates of a common replicate number were, therefore, were calculated relative to *H. melanurus*. The mean optimal number of replicates (averaged across habitat and locality) for different transect dimensions for each species were high, especially for less dominant species, e.g. *S. strigiventer* and *H. chloropterus* (Table 3.6). When averaged across all species, the common optimal mean (as expected) was high even with lower levels of precision (high PV). At a PV of 0.3, the compromise optimal mean varied from 14 replicates for 5 x 100 $m^2$ transect (largest) to 26 replicates for 2 x 30 $m^2$ transect (smallest), which was not feasible for routine sampling. When the abundance (weight) of each species were taken in to account, the (adjusted) mean of optimal number of replicates for less common species was lower. Thus, the compromise optimal mean also decreased, with the 5 x 100 $m^2$ transect (largest) requiring 5 replicates and the 2 x 30 $m^2$ transect requiring 10 replicates for a PV of 0.2, making them realistic options.

The compromise among the optimal means for each species was used to show the relationships between number of replicates and transect dimensions (Fig. 3.3A), and between sampling time and transect dimensions (Fig. 3.3B). It is clear that shorter transects required a greater number of replicates than longer transects, but needed less sampling time. However, there were no clear differences in the number of replicates and sampling time required by different transect widths.

Overall, the results show that a 5 x 30 $m^2$ transect is a good compromise as the optimal transect dimension for estimates abundance of the four wrasse species at the two locations and in the three habitats sampled. At the precision level of 0.2, the adjusted mean optimal number of replicates for estimates *H. melanurus, H. chloropterus, T. lunare* and *S. strigiventer* were 15, 6, 8, and 4 replicates respectively. Thus, the compromise mean optimal number of replicates at the precision level of 0.2 was 8.

## 3.4.2 BENTHIC ASSEMBLAGE

### 3.4.2.1 Area cover estimates

The estimation of area cover (%) of eight benthic categories differed among the four transect lengths, but differences were not consistent between the three habitats, or between two localities (Table 3.7, significantly different of locality-habitat-transect length interaction). The effects of sampling technique varied among habitats and between localities. The lack of a significant interaction between sampling technique and transect length indicates that the effect of transect length was consistent, regardless of the sampling technique.

Canonical Discriminant Analysis (CDA) was used to identify the nature of the complex results detected by MANOVA (Fig. 3.4, and see also Appendix 3.5 - 3.12). The effect of sampling technique was shown on the Cattle Bay-Reef slope (CS) and Hazard Bay-Reef flat (HF), while in other habitats/localities estimates of cover were not influenced by varying transect length. At the reef slope of Cattle Bay, the result from 30, 70 and 100 m transects using the LIT (CSL1, CSL3 and CSL4 respectively) method can be distinguished from the 50 m transect at CSL2 and all transect lengths of FDP (CSF-). The benthic categories responsible for this difference were *Acropora* coral, Soft coral, Massive coral and Fire coral. *Acropora* coral was most prevalent at CSL1, Soft coral at CSL3 and CSL4, Massive coral and Fire coral at CSF and CSL2. At the reef flat of Hazard Bay, the FDP transects detected a lower cover of Dead coral than the LIT transects, separating them on the CDA plot.

The effects of transect length were clearly visible for LIT at the reef flat of Cattle Bay and the reef slope of Cattle Bay and for both sampling techniques at the reef slope of Hazard Bay (Fig. 3.4). At the reef flat of Cattle Bay (CFL-), Dead coral was responsible for the difference of 30 m and 50 m from the main group. At the reef slope of Cattle Bay (CSL-), the 30 m transect estimated more *Acropora* coral, 50 m estimates less Dead coral and more Fire coral while 70 and 100 m estimate more Soft coral. For the reef slope of Hazard Bay (HS--), Massive coral and Fire coral were responsible for the difference when longer transect tend to estimate more Massive coral and Fire coral.

### 3.4.2.2 Precision and efficiency

The most precise transect size of both FDP and LIT for benthic area cover estimates varied for all benthic categories (Table 3.8). For all eight of the benthic categories, however, the 100 m transect has the highest frequency of giving the most precise estimation, for both the FDP and LIT methods (their proportions were 19/43 and 17/44 respectively).

Mean sampling times for FDP transects of 30, 50, 70 and 100 m were 6, 10, 14, and 20 minutes respectively. Similarly, the times for LIT were 13, 20, 27, and 37 minutes.

The most efficient sampling scheme for obtaining a precision value of 0.2 at the different localities and habitats varied across all categories (Table 3.9). The highest frequency of the most efficient transect length was, however, 30 m for both the FDP and LIT methods (their proportion were 13/43 and 17/44 respectively).

### 3.4.2.3 Optimal sampling scheme

The proportional cover of each benthic category were as follows: Dead coral-47.4%, Dead coral with algae-4.6%, Acropora coral-3.7%, Massive coral-6.2%, Branching coral-4.4%, Minor living coral component-1.8%, Fire coral-4.7% and Soft coral-15.2%. The original and adjusted optimal number of replicates calculated for the different sampling schemes differed greatly (Table 3.10). For example, the 30 m of FDP at a precision of 0.2 required 5, 82, 57, 55, 69, 67, 59, and 28 replicates for sampling Dead coral, Dead coral with algae, Acropora coral, Massive coral, Branching coral, Minor living coral component, Fire coral and Soft coral respectively while with adjusted required 5, 8, 5, 8, 7, 3, 6, and 9 replicates respectively. When averaged across variables, mean of original and adjusted optimal number of replicates were 53 and 6 replicates respectively. The results between FDP and LIT, in general, gave very similar results in terms of required numbers of replicates.

The overall performance of both the FDP and LIT methods indicated that short transects were usually more efficient than long transects (Fig. 3.5). The shorter transect of FDP required more replicates but less sampling time than longer transects, especially at a PV of 0.1. For LIT, all four transect lengths required the same number of replicates but shorter transects required less sampling time than longer transects, and were shorter transect therefore more efficient. The 30 m long transect, therefore, was most suitable for both FDP and LIT techniques. At the PV of 0.2, 30 m-FDP requires 6 replicates while 30 m-LIT requires 5 replicates but sampling 30 m-FDP requires less time than 30 m-LIT.

### 3.4.3 FISH-HABITAT RELATIONSHIPS

Results from likelihood ratio analysis indicated that canonical variates 2 to 4 of all combinations of the fish-benthos data set were non-significant (canonical correlation are zero). The results of Canonical Correlation Analysis (CCA), therefore, were presented only from canonical variate 1 (Table 3.11). Transect length appeared to be the most important factor influencing the quantitative relationships between fish and benthic assemblages. The 100 m transect was the only length that obtained significant likelihood ratios for all four combinations between transect widths (fish sampling) and sampling techniques (benthos sampling) while the other transect lengths were not significant. For 100m transects, canonical correlation coefficients, explained variance, predictability of fish, and habitat, were also more consistent than other transect lengths.

The influence of transect widths and sampling techniques on the predictability of habitat and fish, could also be seen especially for 100 m transects. For transect widths, sampling fish with 5 m provided higher predictability of habitat on fish than with 2 m transect. Most of results from other transect lengths showed the same trend. For sampling techniques, sampling benthos with FDP provided higher predictability of fish on habitat than with LIT. However, the results from other transect lengths (whether less consistent) usually obtained low predictability of fish for both sampling techniques.

The influence of different sampling schemes on the relationships between variables of fish and habitat components could be considered from canonical structure of each variable and their own canonical variate (Table 3.12). In general, there were high correlation between *Halichoeres melanurus*, *Thalassoma lunare*, Acropora coral and Soft coral. The influence of transect length appeared to dominate over transect width and sampling technique. The 100 m long transect in any combination of transect width and sampling technique obtained relatively consistent results, not only for these four variables but other variables as well. In contrast, shorter transect length provided less consistency in results and sometimes detected different results. For example, sampling with 30 and 50 m transect did not identify *Acropora* coral as potential variables, while the 70 m transect did, but only for the LIT method. A 30 m transect also detected Dead coral as potential variable (whether not consistent) while other transect lengths did not. Minor influence from transect width and sampling technique may be seen from 100 m transect, as the 5 m wide transect provided a higher correlation for *H. melanurus* and *Acropora* coral than the 2 m wide transect. For sampling techniques, the FDP appeared to provide higher value for dead coral, and dead coral with algae than LIT.

## 3.5 DISCUSSION

Transect-based visual techniques are an important tool for ecological studies of coral reefs. For coral reef fish, a visual census along a transect line is the most popular technique (Dawson-Shepherd *et al.*, 1992) while the line intercept technique is generally accepted as suitable for coral reef benthic studies (Mundy, 1991). The two techniques can be combined to describe correlation between fish and components of the habitat. The bias involved with these transect-based techniques is likely to vary with the species or groups of interest (Sale & Sharp, 1983; Mundy, 1991) and locality or habitat (McCormick & Choat, 1987). As yet, there are no transect based techniques which are suitable for all species and all habitats (Fowler, 1987). Because of cost factors, a pilot sampling study is necessary to determine the most efficient sampling scheme to combine high accuracy and precision with low cost (Snedecor & Cochran, 1967; Underwood, 1981). This problem is of greater importance when a quantitative study on the relationships between coral reef fishes and their associated habitat is required. A significant relationship concluded from a study may well exist, but might be invalid due to the questionable quality of the data, confounded by effects of the sampling scheme. Therefore, an acceptable level of precision and accuracy is needed.

This study assessed the most efficient sampling scheme for fish-habitat relationships, by first independently determining the optimum sampling strategy for each group, and secondly, examining the effect of the different sampling methods on the strength and stability of observed correlation between fish and habitat factors.

### 3.5.1 FISH

Transect dimensions are known to have effects on estimates of mean abundance, and the precision that can be achieved for a given sampling effort (Sale & Sharp, 1983; Fowler, 1987; McCormick & Choat, 1987; Sale, 1996). Changes in the width and length of transects can have independent effects, and estimates can be more sensitive to changes in one, compared with the other. In this study, transect width showed more influence on fish abundance estimates than transect length. The effect of width varied between species, habitat and locality, but when effects were observed, narrow transects gave higher estimates. Similarly, Sale & Sharp (1983) found the estimated density of several reef fishes, including a group of labrids, decreased when transect width was increased from 0.5

to 3 m-wide. In contrast, Fowler (1987) found no significant influence of transect width (1, 2 and 3 m) on the density estimates of three butterflyfishes. These studies indicate that the influence of transect width is species dependent with small size, fast moving, and moderately to highly abundant species (e.g. wrasse) apparently more affected than medium sized, slow moving, and low abundance species (butterflyfishes).

Effects of transect width may be due to bias as a result of boundary effects. That is, problems of inclusion and exclusion of borderline fish from the census (as pointed out by Andrew & Mapstone, 1987). The boundary effects can produce both over and under estimates, depending on species mobility and behaviour. For example, fast moving and wide home range cause higher bias than slow moving and site attach moving species. Thus, the conclusion that the highest estimate is the most accurate (Sale & Sharp, 1983) is not always true. A narrow transect may fail to detect less common and fast moving species especially when estimating in conjunction with a short transect (e.g. *H. chloropterus* and *T. lunare*). A wide transect may underestimate the abundance of juvenile because it is wider than the habitat of a particular fish. This is important for juveniles that are small and often cryptically coloured or hide in a topographically complex habitat (e.g. *T. lunare* and *Dischistodus perspicillatus* - Green, 1992). In general, therefore, wide transects have been selected for sampling large or adult fish while narrow transect are often selected for small cryptic or juvenile fish (Fowler, 1987; Roberts & Ormond, 1987). This may also related to fish abundance and distribution as juveniles are more abundance than adults.

When considering Precision Value, 5 m wide transects appear to give more precise estimates than 2 m wide transects, for the same sampling effort. Since the sampling effort required by both transect widths was similar, 5 m wide transects are likely to be more efficient. Overall, in this case 5 m wide transects are preferable to 2 m-wide transects.

There was little or no influence of transect length on the abundance estimates of these wrasse species. Likewise, Fowler (1987) found that transect length did not significantly influence the estimates of butterflyfishes densities, and McCormick & Choat (1987) found that estimates of abundance of a temperate fish were relatively insensitive to changes in transect length. Unlike transect width, boundary effects and inclusion-exclusion problems are not sources of bias for transect length. Brock (1982) mentioned that short transects allowed sampling within a particular habitat whereas long transects would be more likely to sample a number of habitats. The heterogeneity of habitats within each study area should therefore have some influence on the transect length if within-habitat variability in density is a question of interest. The differences between transect lengths within the range 30 - 100 m is likely to have little influence on wrasse abundance estimates.

Precision estimates were also relatively insensitive to changes in transect length. Since shorter transects were much less costly, in terms of sampling effort, the same precision can be achieved for less effort using short transects. In general, a greater number of short transects is preferable to a smaller number of longer transects (Brock, 1982; McCormick & Choat, 1987; Fowler, 1987). The major advantage to using a number of small sampling units is that greater efficiency (i.e. same precision with less effort) (Pringle, 1984; Downing & Anderson, 1985). Therefore, in this case a 30 m long transect (5m - wide) was adopted.

### 3.5.2 HABITAT

Cover estimates of the various benthic organisms occupying the substratum were influenced by both the line transect method and the length of the transects. Differences in sampling efficiency varied among localities and habitats. In contrast to the fish density estimates, area cover data is proportional, and the estimation of one category is likely to interfere the estimation of another. An accurate and precise estimate is, therefore, necessary to overcome this problem. However, precision is entirely independent of accuracy (Thresher & Gunn, 1986). Of the two different techniques compared in this study, LIT gave the absolute estimates on the entire transect while FDP just estimates over that transect. The results of this study suggested that FDP obtained comparable precision estimates.

The influence of different transect length on accuracy is difficult to determine for the whole spectrum of benthic reef life-forms, because the accuracy of each transect length is not independent (Foster *et al.*, 1991; Meese & Tomich, 1992). That is, if one category is over-estimated, some other must be under-estimated. Therefore, the precision of % cover estimates is an alternative way to compare the performance of different sampling schemes.

The precision of both Fixed Density Point (FDP) and Line Intercept Transect (LIT) methods was comparable, but FDP tended to provide a slightly lower precision value than LIT for the same number of replicates. Basically, LIT should be considered as a reference technique for FDP because LIT collects data on a whole range of sampling unit sizes whereas in FDP, the data are collected by point sampling and accuracy is therefore likely to depend on the density of the sampling points (Foster *et al.*, 1991; Dethier *et al.*, 1993) and abundance of focal organisms (Mundy, 1991; Meese & Tomich, 1992; Dethier *et al.*, 1993). This suggested that data from LIT is more reliable than that obtained by FDP.

There are however, some problems with LIT. Mundy (1991) stated that there has been uncertainty concerning the ability of LIT to provide precise area cover estimates and these factors include observer, transect length and spatial variability of substrata. In this study, spatial variability seems to play an important role in the precision of the area cover estimates as both LIT and FDP displayed the same trends in precision, which varied with habitat and locality. As mentioned earlier, another problem may come from the nature of the data in that each variable (benthic category) is not independent.

A number of studies on sessile benthic organisms have been based on random point sampling (Bainbridge & Reichelt, 1988; Foster *et al.*, 1991, Meese & Tomich, 1992; Santos, 1993 and Dethier *et al.*, 1993; Kaly & Jones, 1994). Foster *et al.* (1991) compared point and photographic quadrat methods of estimating area cover (of sessile marine organisms) and stated that the point quadrat technique had fewer potential biases. Foster *et al.* (1991) also mentioned that the point quadrat technique gave a more accurate estimate of biological cover in layered assemblages but less precision than the photographic technique. Meese & Tomich (1992) compared the results of five techniques for the estimation of benthic area cover (visual, evenly space dots, random dots, stratified random dots and electronic digitizing of photographic images) and found no significant difference between them. Meese & Tomich (1992) also mentioned that random point quadrats frequently failed to detect species covering less than 1% of the sampling unit area. Dethier *et al.* (1993) demonstrated that random point quadrat method is less repeatable and less accurate than visual estimates. Dethier *et al.* (1993) also mentioned that random point quadrats using 100 points were more accurate and less variable than those using 50 points, which often miss rare species of less than 2%. It can therefore be said that the inherent biases of point-based techniques is under and over estimation due to the inability or ability to detect rare components respectively. The degree of bias is directly dependent on the density of points used in sampling. Therefore, it is also necessary to optimise the number of points used in the sampling which might be considered from average colony size of target categories.

A density of one point per meter was used in the FDP in this study. Therefore, benthic lifeforms that had a total area cover less than one meter (less than 1%) would be usually go undetected which short transect likely to have more problem than longer one. This problem was circumvented in this study by using eight major benthic lifeforms derived from grouping a wide division of 14 life-forms. Thus, no categories were very rare. When a systematic sampling technique like FDP was used, the estimates tend to be the same as in the LIT. If more details of the benthic assemblages were needed, FDP may not suitable as its inherent bias estimate on rare categories. The accuracy and precision of FDP can be improved by increasing the density of sampling points. But it is necessary to aware that increasing number of points is directly increasing sampling time (cost).

The influence of transect length of both FDP and LIT varied across localities and habitats and depend on benthic lifeforms (Fig. 3.4). The precision of estimates showed the same major trends. However, with the same number of replicates (4) longer transects provide better precise estimates than short transects. This is because longer transects cover much more distance than shorter transects, and thus have more ability to detect rare categories. Sampling with long transects takes more time and is usually less efficient after optimisation than short transect. hence, there is a trade-off between efficient sampling for common species (short transects better) and the ability to detect uncommon species (long transects better)

### 3.5.3 TRANSECT DIMENSION FOR MULTIPLE SPECIES CENSUSES

An attempt was made to demonstrate the possible ways to optimise sampling scheme when targeting several species, locations and habitats, which has seldom been done. The results indicated that each species or life-form had a different abundance and pattern of distribution across each habitat and locality. These differences had a direct bearing on the calculation of optimal number of replicates. Two categories of abundance, absent and rarely present (discussed below), may wrongly influence the choice of an appropriate transect dimension and provide wrong optimal number of replicates for sampling multiple species at multiple locations.

The decision as to the most efficient sampling scheme is based on the lowest time cost (CT) to achieve a desired level of precision. Time cost is calculated from the mean sampling time (T) of that particular sampling scheme and the optimal number of replicates which is estimated from sample variance. In general, the mean optimal number of replicates for each species can be determined by averaging the optimal number of replicates across all three habitats of both localities. This method can only be used when the distribution pattern of focal groups is not highly variable. However, when the distribution patterns are quite different, including absent and rarely present for some target groups at some habitats or localities, such a mean value may be misleading. In the case where no fishes are present or sampling failing to detect fishes, mean, variance and optimal number of replicates are zero. If these values are used to calculate the mean of optimal number of replicates, it will be an underestimate because a zero precision value can mean highest precision. The number of replicates used in the pilot study may fail to detect rare species. However, if it is necessary to detect rare species, the highest possible replication may be necessary. The effect of absent fish should be considered in the same way as those rarely present. When fishes are rarely present (presence of only one individual at only one time from all replicate) Precision Values calculated from these samples tend to be one (least precise estimates). This means the optimal number of replicates needed for sampling these rare fishes must be the maximum possible number of replicates for a particular precision value.

In the literature, precision of estimates tend to be assessed only from dominant or common categories (Fig. 3.6). Similar trends for different transect length (results not shown) suggest that the relationships between area cover and precision value are logarithmic. Note that the cost of sampling a transect will exponentially increase with increasing coral cover/density. Rare categories tended to produce very high precision values (imprecise estimate). An area cover of less than 3% will tend to produce a precision value higher than 0.5, which results in a very high optimal number of replicates. This high number of replicates is generally impractical both in terms of single variable or multiple categories or species. Mundy (1991) also found the same trend and mentioned that species which occupied less than one percent of the total area tended to produce very high precision values. This trend was also reported for coral reef fish (Meekan *et al.*, 1995).

The classification level of benthic life-forms used are limited by their relative abundance which is also related directly to the numbers of category (species). For example, Fig. 3.6 illustrates that when area cover less than 5% the precision value tended to increase very rapidly. If 5% cover is used as the lower limit for % cover data and assuming all categories equally common, the number of life-forms used should not exceed 20 life-forms which is difficult if collecting data at species level. However, relative abundance of variables are rarely the same in the real situations. Without weight adjusting to compensate for relative abundance, optimisation results for sampling all/most categories/species in the assemblage may be impractical due to the influence of less common species/categories (Table 3.6 and 3.10). Therefore, the decision by considering how the abundance of each species contribute to the assemblage (weight) is an alternative.

To optimise the sampling unit number, therefore, the aggregation of relevant variables into larger categories is important and should be done first. Adjusting optimal number of replicates of each categories by applied different weights can then be applied.

### 3.5.4 SAMPLING SCHEMES AND FISH-HABITAT RELATIONSHIPS

The quantitative relationship between fishes and habitat was directly dependent on the sampling scheme applied in this study. Similarly, Roberts & Ormond (1987) applied different transect dimensions for sampling fish assemblages from adjacent areas and also detected some different quantitative relationships between the fish community parameters and substratum variables. However, in their case results may also be confounded by sampling technique in which fish and habitat variables were not all collected on the same transect. The results of this study show that the choice of sampling scheme is significant and may lead to incorrect conclusions if the sampling scheme is not appropriate to the objectives of the study.

It is very difficult to assess the true (accurate) relationships between two variables from observation as the true values of both variables are unknown. Determination of precision is the only realistic guideline. The results of this study showed that high precision sampling schemes appeared to produce consistent results, while low-precision sampling schemes were likely to provide non-consistent results and even contrasting patterns. Transect dimension appeared to be the most important factor for sampling fish and more important than technique for sampling habitat. This result related directly to the sampling precision of each factors (discuss earlier). The optimal sampling scheme for each variable will also provide the most precise estimates for their relationship, for an equivalent cost. Note that sampling for the study of relationships should be done on the same area (transect).

A 30 m transect length was the most preference for both habitat structure and fish abundance sampling. For habitat structure, both FDP and LIT illustrated comparable results. FDP needed less sampling time but there was an inherent bias when need to study more details of the assemblage. At the PV of 0.2, FDP required 6 replicates for a total of 36 minutes while LIT required 5 replicates and a total of 65 minutes. The FDP was therefore preferred when studying major benthic life-forms. For a more detailed study, LIT was preferred because of the tendency in FDP to the bias estimates (over and under) of less common and rare life-forms. For fish abundance, transect of dimensions 5x30 $m^2$ was preferred which needed 8 replicates for a PV of 0.2. Overall, the preferred sampling scheme will rely on 30m transect with 7 replicates.

### 3.5.5 CONCLUSIONS

It is important to optimise sampling scheme as it will provide the required data quality at a reasonable cost. Optimising a sampling scheme with respect to cost efficiency has primarily been restricted to single species at a single location. The application for multiple species at multi-spatial scales clearly needs additional procedures. Here I suggested one approach, based on using the difference in the abundance of each species as a factor in weighting estimates of optimal replication for each species to arrive at a compromise figure. However, there is an influence of spatial variation on the precision of estimates, which varies depending on distribution and abundance of species. The different sampling schemes, especially in terms of transect length, therefore, provided data of differing quality. Application of the precision value is very useful in solving this problem. With the same precision level, the optimal number of replicate of different sampling schemes can be calculated. Differences in data quality (from different sampling scheme) will also have an influence on the quantitative relationship between fish and their habitat. However, high quality data (high precision) will provide consistent results. It is important, therefore, to use a consistent sampling scheme throughout a study that aims to investigate the quantitative relationships of two or more components. The optimal sampling scheme for multi-species assemblages also has an advantage in that it is not necessary to repeat optimisation procedures every time there is a focus on a different species or group.

Table 3.1. Wrasses (A) and benthic lifeforms (B) with the associated abbreviations used in this study.

| Species/Group | Abbreviation |
|---|---|
| A) Wrasse | |
| *Halichoeres melanurus* | Hamel |
| *Halichoeres chloropterus* | Hachl |
| *Thalassoma lunare* | Thlun |
| *Stethojulis strigiventer* | Ststr |
| | |
| B) Benthic lifeforms | |
| *Acropora* coral | AC |
| Massive coral | CM |
| Branching coral | CB |
| Minor living coral component | MLC |
| Fire coral | FC |
| Soft coral | SC |
| Other fauna | OT (e.g. sponges, zooanthids, ascidians) |
| Dead coral | DC |
| Dead coral with algae | DCA |
| Abiotics | ABIO (e.g. rock, sand) |

Table 3.2. Combination of transect length, transect width (fish sampling) and sampling technique (habitat sampling) investigated in order to examine their influence on fish-habitat relationships.

| Transect length | Transect width | Sampling technique |
|---|---|---|
| 30 m | 2 m | Fixed Density Point Transect |
| | 5 m | Fixed Density Point Transect |
| | 2 m | Line Intercept Transect |
| | 5 m | Line Intercept Transect |
| 50 m | 2 m | Fixed Density Point Transect |
| | 5 m | Fixed Density Point Transect |
| | 2 m | Line Intercept Transect |
| | 5 m | Line Intercept Transect |
| 70 m | 2 m | Fixed Density Point Transect |
| | 5 m | Fixed Density Point Transect |
| | 2 m | Line Intercept Transect |
| | 5 m | Line Intercept Transect |
| 100 m | 2 m | Fixed Density Point Transect |
| | 5 m | Fixed Density Point Transect |
| | 2 m | Line Intercept Transect |
| | 5 m | Line Intercept Transect |

Table 3.3. MANOVA results showing the influence of four transect lengths, two transect widths on the abundance estimates of four wrasse species: *Halichoeres melanurus, H. chloropterus, Thalassoma lunare* and *Stethojulis strigiventer*. Data included two localities each with three habitats. (* = significant at $P < 0.05$, ns = non-significant)

| Source | DF | Pillai's Trace | F Value | P |
|---|---|---|---|---|
| Locality, S | 4, 141 | 0.57 | 46.15 | <0.001* |
| Habitat, H | 8, 284 | 0.98 | 33.87 | <0.001 * |
| Transect width, W | 4, 141 | 0.26 | 12.38 | <0.001 * |
| Transect length, L | 12, 429 | 0.06 | 0.73 | 0.721 ns |
| SxH | 8, 284 | 0.28 | 5.77 | <0.001 * |
| SxW | 4, 141 | 0.06 | 2.21 | 0.070 ns |
| SxL | 12, 429 | 0.12 | 1.53 | 0.109 ns |
| HxW | 8, 284 | 0.14 | 2.60 | 0.009 * |
| HxL | 24, 576 | 0.11 | 0.70 | 0.855 ns |
| WxL | 12, 429 | 0.02 | 0.30 | 0.990 ns |
| SxHxW | 8, 284 | 0.16 | 3.00 | 0.003 * |
| SxHxL | 24, 576 | 0.08 | 0.51 | 0.974 ns |
| SxWxL | 12, 429 | 0.04 | 0.54 | 0.892 ns |
| HxWxL | 24, 576 | 0.09 | 0.52 | 0.971 ns |
| SxHxWxL | 24, 576 | 0.06 | 0.36 | 0.998 ns |

Table 3.4. Transect dimensions (width x length) which provided the most precise abundance estimates for each of four wrasses at different habitats (Reef flat, Reef edge, Reef slope) and localities ( CB = Cattle Bay, HB = Hazard Bay).

| Habitat | Reef flat | | Reef edge | | Reef slope | |
|---|---|---|---|---|---|---|
| Locality | CB | HB | CB | HB | CB | HB |
| *H. melanurus* | 2x70 | 5x50 | 5x30 | 5x30 | 5x100 | 2x70 |
| *H. chloropterus* | 5x100 | 2x50 | 5x100 | 5x70 | 5x100 | 5x70 or 100 |
| *T. lunare* | 5x100 | 5x100 | 2x100 | 5x70 | 5x100 | 5x100 |
| *S. strigiventer* | 5x100 | 5x50 | 2x100 | 2x30 | 2x50 | 2 or 5x30 |

Table 3.5. The most efficient transect dimensions (width x length) at Precision value = 0.2 found for the estimation of abundance of four wrasses at different habitats and localities (as in Table 3.4).

| Habitat | Reef flat | | Reef edge | | Reef slope | |
|---|---|---|---|---|---|---|
| Locality | CB | HB | CB | HB | CB | HB |
| *H. melanurus* | 5x30 | 5x30 | 5x30 | 5x30 | 5x30 | 5x30 |
| *H. chloropterus* | 5x100 | 5x30 | 2x30 | 5x50 | 2x50 | 5x70 |
| *T. lunare* | 5x30 | 5x70 | 5x30 | 5x30 | 5x30 | 5x100 |
| *S. strigiventer* | 5x30 | 5x30 | 2x30 | 2x50 | 2x50 | 2x30 |

Table 3.6. Original mean and adjusted mean (original mean x weighting) of optimal number of replicates at three precision levels: 0.1, 0.2, and 0.3, for abundance estimates of four wrasses; *Halichoeres melanurus*, *H. chloropterus*, *Thalassoma lunare*, and *Stethojulis strigiventer*.

| Precision level | Transect Width | Transect Length | Species | *H. melanurus* | *H. chloropterus* | *T. lunare* | *S. strigiventer* | MEAN |
|---|---|---|---|---|---|---|---|---|
| | | | Proportion | 0.764 | 0.076 | 0.121 | 0.039 | |
| | | | Weight | 1 | 0.099 | 0.158 | 0.051 | |
| PV 0.1 | 2m | 30m | Original mean | 63.70 | 309.78 | 249.38 | 307.41 | 233 |
| | | | Adjusted mean | 63.70 | 30.67 | 39.40 | 15.68 | 38 |
| | | 50m | Original mean | 67.68 | 216.87 | 327.04 | 261.33 | 219 |
| | | | Adjusted mean | 67.68 | 21.47 | 51.67 | 13.33 | 39 |
| | | 70m | Original mean | 43.45 | 237.19 | 250.27 | 272.44 | 201 |
| | | | Adjusted mean | 43.45 | 23.48 | 39.54 | 13.89 | 31 |
| | | 100m | Original mean | 41.04 | 200.81 | 215.33 | 279.01 | 185 |
| | | | Adjusted mean | 41.04 | 19.88 | 34.02 | 14.23 | 28 |
| | 5m | 30m | Original mean | 58.12 | 231.50 | 189.18 | 279.56 | 190 |
| | | | Adjusted mean | 58.12 | 22.92 | 29.89 | 14.26 | 32 |
| | | 50m | Original mean | 27.22 | 140.07 | 167.16 | 253.64 | 148 |
| | | | Adjusted mean | 27.22 | 13.87 | 26.41 | 12.94 | 21 |
| | | 70m | Original mean | 43.09 | 111.82 | 153.07 | 229.78 | 135 |
| | | | Adjusted mean | 43.09 | 11.07 | 24.18 | 11.72 | 23 |
| | | 100m | Original mean | 31.36 | 71.95 | 148.89 | 220.32 | 119 |
| | | | Adjusted mean | 31.36 | 7.12 | 23.52 | 11.24 | 19 |
| PV 0.2 | 2m | 30m | Original mean | 15.92 | 77.44 | 62.35 | 76.85 | 59 |
| | | | Adjusted mean | 15.92 | 7.67 | 9.85 | 3.92 | 10 |
| | | 50m | Original mean | 16.92 | 54.22 | 81.76 | 65.33 | 55 |
| | | | Adjusted mean | 16.92 | 5.37 | 12.92 | 3.33 | 10 |
| | | 70m | Original mean | 10.86 | 59.30 | 62.57 | 68.11 | 51 |
| | | | Adjusted mean | 10.86 | 5.87 | 9.89 | 3.47 | 8 |
| | | 100m | Original mean | 10.26 | 50.20 | 53.83 | 69.75 | 47 |
| | | | Adjusted mean | 10.26 | 4.97 | 8.51 | 3.56 | 7 |
| | 5m | 30m | Original mean | 14.53 | 57.87 | 47.30 | 69.89 | 48 |
| | | | Adjusted mean | 14.53 | 5.73 | 7.47 | 3.56 | 8 |
| | | 50m | Original mean | 6.80 | 35.02 | 41.79 | 63.41 | 37 |
| | | | Adjusted mean | 6.80 | 3.47 | 6.60 | 3.23 | 6 |
| | | 70m | Original mean | 10.77 | 27.96 | 38.27 | 42.44 | 30 |
| | | | Adjusted mean | 10.77 | 2.77 | 6.05 | 2.16 | 6 |
| | | 100m | Original mean | 7.84 | 17.99 | 37.22 | 55.08 | 30 |
| | | | Adjusted mean | 7.84 | 1.78 | 5.88 | 2.81 | 5 |
| PV 0.3 | 2m | 30m | Original mean | 7.08 | 34.51 | 27.71 | 34.15 | 26 |
| | | | Adjusted mean | 7.08 | 3.42 | 4.38 | 1.74 | 5 |
| | | 50m | Original mean | 7.52 | 24.10 | 36.34 | 29.04 | 25 |
| | | | Adjusted mean | 7.52 | 2.39 | 5.74 | 1.48 | 5 |
| | | 70m | Original mean | 4.83 | 26.35 | 27.81 | 30.36 | 23 |
| | | | Adjusted mean | 4.83 | 2.61 | 4.39 | 1.55 | 4 |
| | | 100m | Original mean | 4.56 | 22.31 | 23.92 | 31.09 | 21 |
| | | | Adjusted mean | 4.56 | 2.21 | 3.78 | 1.59 | 4 |
| | 5m | 30m | Original mean | 6.46 | 25.72 | 21.02 | 31.06 | 22 |
| | | | Adjusted mean | 6.46 | 2.55 | 3.32 | 1.58 | 4 |
| | | 50m | Original mean | 3.02 | 15.56 | 18.57 | 28.27 | 17 |
| | | | Adjusted mean | 3.02 | 1.54 | 2.93 | 1.44 | 3 |
| | | 70m | Original mean | 4.79 | 12.42 | 17.01 | 25.62 | 15 |
| | | | Adjusted mean | 4.79 | 1.23 | 2.69 | 1.31 | 3 |
| | | 100m | Original mean | 3.48 | 7.99 | 16.54 | 24.57 | 14 |
| | | | Adjusted mean | 3.48 | 0.79 | 2.61 | 1.25 | 3 |

Table 3.7. MANOVA results showing the influence of two sampling techniques and four transect lengths on the estimates of eight benthic categories: *Acropora* coral, Massive coral, Branching coral, Minor living coral component, Fire coral, Soft coral, Dead coral, and Dead coral with algae, at two localities each with three habitats (* = significant at P <0.05, ns = non-significant).

| Source | df | Pillai's trace | F-value | *P* |
|---|---|---|---|---|
| Locality, A | 8, 137 | 0.59 | 24.41 | <0.001* |
| Habitat, B | 16, 276 | 1.35 | 35.62 | <0.001* |
| A*B | 16, 276 | 0.86 | 13.05 | <0.001* |
| Technique, C | 8, 137 | 0.17 | 3.46 | 0.001* |
| A*C | 8, 137 | 0.14 | 2.85 | 0.006* |
| B*C | 16, 276 | 0.23 | 2.25 | 0.004* |
| A*B*C | 16, 276 | 0.12 | 1.23 | $0.330^{ns}$ |
| Transect Length, D | 24, 417 | 0.32 | 2.06 | 0.003* |
| A*D | 24, 417 | 0.31 | 1.96 | 0.005* |
| B*D | 48, 852 | 0.49 | 1.59 | 0.008* |
| A*B*D | 48, 852 | 0.54 | 1.74 | 0.002* |
| C*D | 24, 417 | 0.15 | 0.91 | $0.590^{ns}$ |
| A*C*D | 24, 417 | 0.09 | 0.53 | $0.968^{ns}$ |
| B*C*D | 48, 852 | 0.33 | 1.04 | $0.402^{ns}$ |
| A*B*C*D | 48, 852 | 0.35 | 1.09 | $0.317^{ns}$ |

Table 3.8. The most precise transect lengths (meter) for the two sampling techniques for area cover estimation for the eight benthic categories at different habitats and localities: * cannot detect, DC = Dead Coral, DCA = Dead Coral with algae, AC = Acropora coral, CM = Massive Coral, CB = Branching Coral, MLC = Minor component of Living Coral, FC = Fire Coral, SC = Soft Coral.

| | | | Major benthic life-forms | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| Technique | locality | Habitat | AC | CM | CB | MLC | FC | SC | DC | DCA |
| FDP | Cattle Bay | Flat | 50 | 50 | 30 | * | * | 50 | 70 | 100 |
| | | Edge | 30 | 100 | 100 | 100 | 100 | 50 | 50 | 100 |
| | | Slope | 100 | 100 | 50 | 100 | 50 | 30 | 50 | 30 |
| | Hazard Bay | Flat | 70 | 100 | 100 | 30 | * | 70 | 30 | * |
| | | Edge | 100 | 100 | 70 | 70 | 100 | 50 | 70 | * |
| | | Slope | 70 | 100 | 70 | 100 | 100 | 100 | 100 | 100 |
| LIT | Cattle Bay | Flat | 50 | 30 | 30 | 30 | * | 50 | 50 | 70 |
| | | Edge | 100 | 50 | 50 | 100 | 30 | 100 | 50 | 100 |
| | | Slope | 30 | 100 | 100 | 70 | 50 | 30 | 50 | 50 |
| | Hazard Bay | Flat | 100 | 70 | 100 | 70 | * | 100 | 70 | 100 |
| | | Edge | 70 | 100 | 50 | 70 | 100 | 100 | 100 | * |
| | | Slope | 70 | 70 | 70 | 100 | 100 | 30 | 100 | * |

Table 3.9. The most efficient transect length (meter) for both sampling techniques for area cover estimation for the eight benthic lifeforms at different habitats and localities: * = cannot detect benthic lifeforms, benthic lifeform abbreviations as in Table3.2.

| | | | Major benthic life-forms | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| Technique | locality | Habitat | AC | CM | CB | MLC | FC | SC | DC | DCA |
| FDP | Cattle Bay | Flat | 50 | 50 | 30 | * | * | 50 | 70 | 30 |
| | | Edge | 30 | 50 | 50 | 30 | 30 | 50 | 70 | 30 |
| | | Slope | 30 | 50 | 50 | 30 | 50 | 50 | 50 | 30 |
| | Hazard Bay | Flat | 70 | 100 | 100 | 30 | * | 30 | 30 | * |
| | | Edge | 100 | 100 | 30 | 70 | 100 | 50 | 70 | * |
| | | Slope | 70 | 100 | 70 | 100 | 100 | 100 | 100 | 100 |
| LIT | Cattle Bay | Flat | 50 | 30 | 30 | * | * | 50 | 30 | 70 |
| | | Edge | 30 | 50 | 50 | 30 | 30 | 100 | 50 | 30 |
| | | Slope | 30 | 50 | 30 | 70 | 50 | 30 | 50 | 30 |
| | Hazard Bay | Flat | 30 | 30 | 100 | 70 | * | 30 | 30 | 100 |
| | | Edge | 70 | 100 | 50 | 70 | 100 | 70 | 100 | * |
| | | Slope | 70 | 70 | 30 | 100 | 100 | 30 | 100 | * |

Table 3.10. Original mean and adjusted mean (original mean x weighting) of optimal number of replicates at three precision levels: 0.1, 0.2 and 0.3, for area cover estimates of eight reef benthic lifeforms. Abbreviation of reef benthic lifeforms as in Table 3.1.

| Precision level | Technuque | Transect Length | Lifeforms | DC | DCA | AC | CM | CB | MLC | FC | SC | MEAN |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | Proportion | 47.35 | 4.61 | 3.74 | 6.20 | 4.30 | 1.80 | 4.68 | 15.16 | |
| | | | Weight | 1.00 | 0.10 | 0.08 | 0.13 | 0.09 | 0.04 | 0.10 | 0.32 | |
| PV 0.1 | FDP | 30 | Original | 15.3 | 324.7 | 226.2 | 215.3 | 274.8 | 265.5 | 232.0 | 51.8 | 201 |
| | | | Adjusted | 15.3 | 31.6 | 17.9 | 28.2 | 25.0 | 10.1 | 22.9 | 16.6 | 21 |
| | | 50 | Original | 11.0 | 400.0 | 245.7 | 94.3 | 216.7 | 317.8 | 212.0 | 31.2 | 192 |
| | | | Adjusted | 11.0 | 38.9 | 19.4 | 12.4 | 19.7 | 12.1 | 21.0 | 10.0 | 19 |
| | | 70 | Original | 9.0 | 331.0 | 69.2 | 65.8 | 228.0 | 243.8 | 179.3 | 59.5 | 149 |
| | | | Adjusted | 9.0 | 32.2 | 5.5 | 8.6 | 20.7 | 9.3 | 17.7 | 19.1 | 16 |
| | | 100 | Original | 7.2 | 297.2 | 108.5 | 51.3 | 195.2 | 222.5 | 164.5 | 58.0 | 139 |
| | | | Adjusted | 7.2 | 28.9 | 8.6 | 6.7 | 17.7 | 8.5 | 16.3 | 18.6 | 15 |
| | LIT | 30 | Original | 12.2 | 314.7 | 152.2 | 83.0 | 239.7 | 321.2 | 226.5 | 38.6 | 174 |
| | | | Adjusted | 12.2 | 30.6 | 12.0 | 10.9 | 21.8 | 12.2 | 22.4 | 12.4 | 17 |
| | | 50 | Original | 6.2 | 400.0 | 142.5 | 52.5 | 206.3 | 269.3 | 215.3 | 92.7 | 174 |
| | | | Adjusted | 6.2 | 38.9 | 11.3 | 6.9 | 18.7 | 10.2 | 21.3 | 29.7 | 18 |
| | | 70 | Original | 18.7 | 298.0 | 84.2 | 46.3 | 210.5 | 220.5 | 184.5 | 91.5 | 145 |
| | | | Adjusted | 18.7 | 29.0 | 6.6 | 6.1 | 19.1 | 8.4 | 18.2 | 29.3 | 17 |
| | | 100 | Original | 16.8 | 303.3 | 104.7 | 35.3 | 134.3 | 187.7 | 163.8 | 68.3 | 127 |
| | | | Adjusted | 16.8 | 29.5 | 8.3 | 4.6 | 12.2 | 7.1 | 16.2 | 21.9 | 15 |
| PV 0.2 | FDP | 30 | Original | 4.3 | 81.2 | 56.7 | 54.2 | 69.0 | 66.5 | 58.2 | 27.7 | 53 |
| | | | Adjusted | 4.3 | 7.9 | 4.5 | 7.1 | 6.3 | 2.5 | 5.7 | 8.9 | 6 |
| | | 50 | Original | 3.2 | 100.0 | 61.5 | 23.7 | 54.3 | 79.5 | 53.2 | 23.5 | 50 |
| | | | Adjusted | 3.2 | 9.7 | 4.9 | 3.1 | 4.9 | 3.0 | 5.3 | 7.5 | 6 |
| | | 70 | Original | 2.8 | 82.8 | 17.7 | 16.8 | 57.2 | 61.2 | 45.0 | 15.2 | 38 |
| | | | Adjusted | 2.8 | 8.1 | 1.4 | 2.2 | 5.2 | 2.3 | 4.4 | 4.9 | 4 |
| | | 100 | Original | 2.3 | 74.7 | 27.5 | 13.2 | 49.0 | 55.8 | 41.5 | 14.7 | 35 |
| | | | Adjusted | 2.3 | 7.3 | 2.2 | 1.7 | 4.4 | 2.1 | 4.1 | 4.7 | 4 |
| | LIT | 30 | Original | 3.7 | 78.7 | 38.5 | 21.0 | 60.2 | 80.3 | 56.8 | 25.2 | 46 |
| | | | Adjusted | 3.7 | 7.7 | 3.0 | 2.7 | 5.5 | 3.1 | 5.6 | 8.1 | 5 |
| | | 50 | Original | 2.0 | 100.0 | 35.8 | 13.5 | 52.0 | 67.5 | 54.0 | 23.7 | 44 |
| | | | Adjusted | 2.0 | 9.7 | 2.8 | 1.8 | 4.7 | 2.6 | 5.3 | 7.6 | 5 |
| | | 70 | Original | 5.2 | 74.8 | 21.5 | 12.0 | 52.8 | 55.3 | 46.3 | 23.0 | 37 |
| | | | Adjusted | 5.2 | 7.3 | 1.7 | 1.6 | 4.8 | 2.1 | 4.6 | 7.4 | 5 |
| | | 100 | Original | 4.7 | 76.0 | 26.5 | 9.2 | 48.8 | 47.2 | 41.2 | 17.3 | 34 |
| | | | Adjusted | 4.7 | 7.4 | 2.1 | 1.2 | 4.4 | 1.8 | 4.1 | 5.5 | 4 |
| PV 0.3 | FDP | 30 | Original | 2.3 | 36.7 | 25.7 | 24.3 | 31.2 | 30.2 | 26.3 | 12.5 | 24 |
| | | | Adjusted | 2.3 | 3.6 | 2.0 | 3.2 | 2.8 | 1.1 | 2.6 | 4.0 | 3 |
| | | 50 | Original | 1.8 | 45.0 | 27.8 | 10.8 | 24.7 | 35.8 | 24.2 | 10.8 | 23 |
| | | | Adjusted | 1.8 | 4.4 | 2.2 | 1.4 | 2.2 | 1.4 | 2.4 | 3.5 | 3 |
| | | 70 | Original | 1.5 | 37.3 | 8.2 | 7.7 | 26.0 | 27.7 | 20.5 | 7.0 | 17 |
| | | | Adjusted | 1.5 | 3.6 | 0.6 | 1.0 | 2.4 | 1.1 | 2.0 | 2.2 | 2 |
| | | 100 | Original | 1.5 | 33.5 | 12.5 | 6.3 | 22.2 | 25.3 | 18.8 | 6.8 | 16 |
| | | | Adjusted | 1.5 | 3.3 | 1.0 | 0.8 | 2.0 | 1.0 | 1.9 | 2.2 | 2 |
| | LIT | 30 | Original | 1.8 | 35.3 | 17.3 | 9.7 | 27.2 | 36.2 | 25.7 | 11.3 | 21 |
| | | | Adjusted | 1.8 | 3.4 | 1.4 | 1.3 | 2.5 | 1.4 | 2.5 | 3.6 | 3 |
| | | 50 | Original | 1.2 | 45.0 | 16.3 | 6.5 | 23.3 | 30.5 | 24.5 | 10.7 | 20 |
| | | | Adjusted | 1.2 | 4.4 | 1.3 | 0.9 | 2.1 | 1.2 | 2.4 | 3.4 | 3 |
| | | 70 | Original | 2.5 | 33.5 | 9.8 | 5.7 | 24.0 | 25.2 | 21.0 | 10.7 | 17 |
| | | | Adjusted | 2.5 | 3.3 | 0.8 | 0.7 | 2.2 | 1.0 | 2.1 | 3.4 | 2 |
| | | 100 | Original | 2.5 | 34.0 | 12.0 | 4.5 | 22.2 | 21.3 | 18.7 | 7.8 | 16 |
| | | | Adjusted | 2.5 | 3.3 | 0.9 | 0.6 | 2.0 | 0.8 | 1.8 | 2.5 | 2 |

Table 3.11. Summarised results of the first canonical variate from canonical correlation analysis showing variation in quantitative relationships between fish and benthic (habitat) assemblages using different sampling schemes. 2 m = 2 m wide transect for fish sampling, 5 m = 5 m wide transect for fish sampling, FDP = Fixed Density Point transect for benthos sampling, LIT = Line Intercept Transect for benthos sampling, * = significant at $P < 0.05$, and $^{ns}$ = non-significant.

| Source/Transect length | 30 m | 50 m | 70 m | 100 m |
|---|---|---|---|---|
| 1) Canonical correlation | | | | |
| 2 m – FDP | 0.89 | 0.91 | 0.86 | 0.95 |
| 5 m – FDP | 0.92 | 0.91 | 0.88 | 0.91 |
| 2 m – LIT | 0.91 | 0.87 | 0.88 | 0.92 |
| 5 m – LIT | 0.83 | 0.78 | 0.86 | 0.89 |
| 2) Explained variance (%) | | | | |
| 2 m - FDP | 76 | 76 | 69 | 74 |
| 5 m - FDP | 70 | 64 | 72 | 67 |
| 2 m - LIT | 75 | 70 | 65 | 67 |
| 5 m - LIT | 62 | 60 | 57 | 62 |
| 3) P-value of likelihood ratio | | | | |
| 2 m - FDP | $0.636^{ns}$ | 0.066* | $0.076^{ns}$ | 0.0003* |
| 5 m - FDP | 0.002* | 0.004* | $0.138^{ns}$ | 0.008* |
| 2 m - LIT | 0.049* | $0.052^{ns}$ | 0.047* | 0.008* |
| 5 m - LIT | $0.338^{ns}$ | $0.348^{ns}$ | $0.060^{ns}$ | 0.024* |
| 4) Fish predictability (%) | | | | |
| 2 m - FDP | 31 | 13 | 12 | 19 |
| 5 m - FDP | 14 | 11 | 18 | 24 |
| 2 m - LIT | 15 | 7 | 11 | 8 |
| 5 m - LIT | 29 | 15 | 9 | 8 |
| 5) Habitat predictability (%) | | | | |
| 2 m - FDP | 29 | 35 | 31 | 38 |
| 5 m - FDP | 39 | 33 | 39 | 45 |
| 2 m - LIT | 28 | 27 | 33 | 36 |
| 5 m - LIT | 31 | 19 | 37 | 45 |

Table 3.12. Canonical structure of fish and habitat variables with their own (first) canonical variate showing variation produced by different 16 sampling schemes. Details of sampling scheme were described in Table 3.2, Hamel = *Halichoeres melanurus*, Hachl = *H. chloropterus*, Thlun = *Thalassoma lunare*, Ststr = *Stethojulis strigiventer*, AC = *Acropora* coral, CM = Massive coral, CB = Branching coral, MLC = Minor living coral component, FC = Fire coral, SC = Soft coral, DC = Dead coral, DCA = Dead coral with algae, bold value representing variable described in text.

| | Fish | | | | Habitat | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Variable<br>Sampling | Hamel | Hachl | Thlun | Ststr | AC | CM | CB | MLC | FC | SC | DC | DCA |
| 1) 30 m transect | | | | | | | | | | | | |
| 2 m – FDP | **0.69** | -0.09 | **0.95** | -0.08 | **0.31** | 0.15 | 0.61 | 0.06 | 0.54 | **0.72** | **-0.79** | 0.24 |
| 5 m – FDP | **0.96** | 0.21 | **0.53** | 0.52 | **0.24** | 0.46 | 0.15 | 0.21 | 0.30 | **0.77** | **-0.17** | -0.17 |
| 2 m – LIT | **0.84** | -0.27 | **0.68** | -0.24 | **0.14** | 0.52 | 0.21 | 0.42 | 0.40 | **0.70** | **-0.34** | -0.09 |
| 5 m - LIT | **0.83** | 0.08 | **0.77** | 0.53 | **0.38** | 0.36 | 0.33 | 0.27 | 0.62 | **0.65** | **-0.82** | 0.34 |
| 2) 50 m transect | | | | | | | | | | | | |
| 2 m - FDP | **0.72** | -0.22 | **0.91** | 0.14 | **0.34** | 0.38 | 0.23 | 0.19 | 0.31 | **0.85** | -0.24 | -0.05 |
| 5 m - FDP | **0.67** | 0.11 | **0.91** | 0.24 | **0.41** | -0.05 | 0.19 | -0.06 | 0.05 | **0.90** | -0.07 | -0.10 |
| 2 m - LIT | **0.78** | -0.02 | **0.80** | 0.20 | **0.35** | 0.11 | 0.03 | 0.33 | 0.16 | **0.60** | -0.14 | 0.00 |
| 5 m - LIT | **0.77** | -0.03 | **0.69** | -0.02 | **0.19** | 0.09 | 0.39 | 0.23 | 0.13 | **0.83** | -0.42 | 0.00 |
| 3) 70 m transect | | | | | | | | | | | | |
| 2 m - FDP | **0.51** | 0.05 | **0.98** | 0.16 | **0.31** | 0.07 | 0.33 | 0.50 | 0.23 | **0.72** | -0.15 | 0.16 |
| 5 m - FDP | **0.82** | 0.21 | **0.84** | 0.57 | **0.70** | -0.05 | 0.08 | 0.42 | 0.03 | **0.73** | -0.39 | 0.41 |
| 2 m - LIT | **0.52** | 0.08 | **0.96** | -0.13 | **0.26** | 0.08 | 0.44 | 0.28 | 0.26 | **0.72** | -0.26 | 0.03 |
| 5 m - LIT | **0.75** | 0.78 | **0.71** | 0.45 | **0.76** | -0.49 | -0.23 | -0.30 | -0.20 | **0.41** | -0.19 | 0.31 |
| 4) 100 m transect | | | | | | | | | | | | |
| 2 m - FDP | **0.70** | 0.17 | **0.87** | 0.32 | **0.73** | 0.03 | 0.12 | 0.18 | -0.02 | **0.72** | -0.35 | 0.27 |
| 5 m - FDP | **0.90** | 0.27 | **0.78** | 0.71 | **0.84** | -0.03 | -0.06 | 0.25 | -0.06 | **0.72** | -0.55 | 0.38 |
| 2 m - LIT | **0.72** | 0.16 | **0.88** | 0.26 | **0.70** | -0.04 | 0.14 | 0.23 | -0.03 | **0.72** | -0.05 | 0.01 |
| 5 m - LIT | **0.92** | 0.42 | **0.83** | 0.51 | **0.79** | 0.26 | 0.00 | 0.11 | -0.13 | **0.67** | -0.03 | 0.01 |

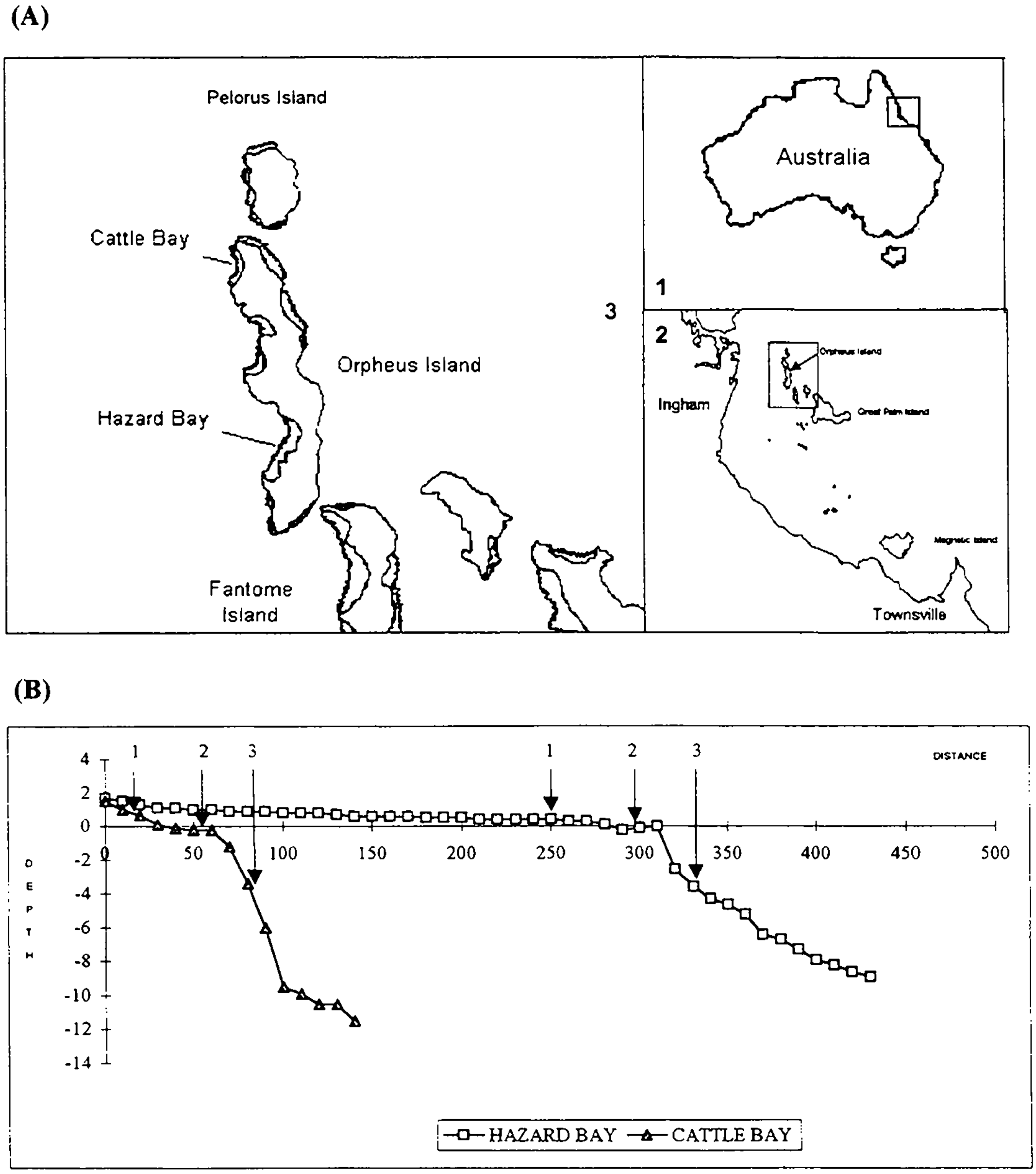


Fig 3.1. Details of the study area: A) Maps showing study localities and B) shore profiles of Cattle Bay and Hazard Bay. Arrows indicate three habitats used: 1= Reef flat, 2=Reef edge, and 3=Reef slope, depth relative to Mean Sea Level, depth and distance scales in metres.

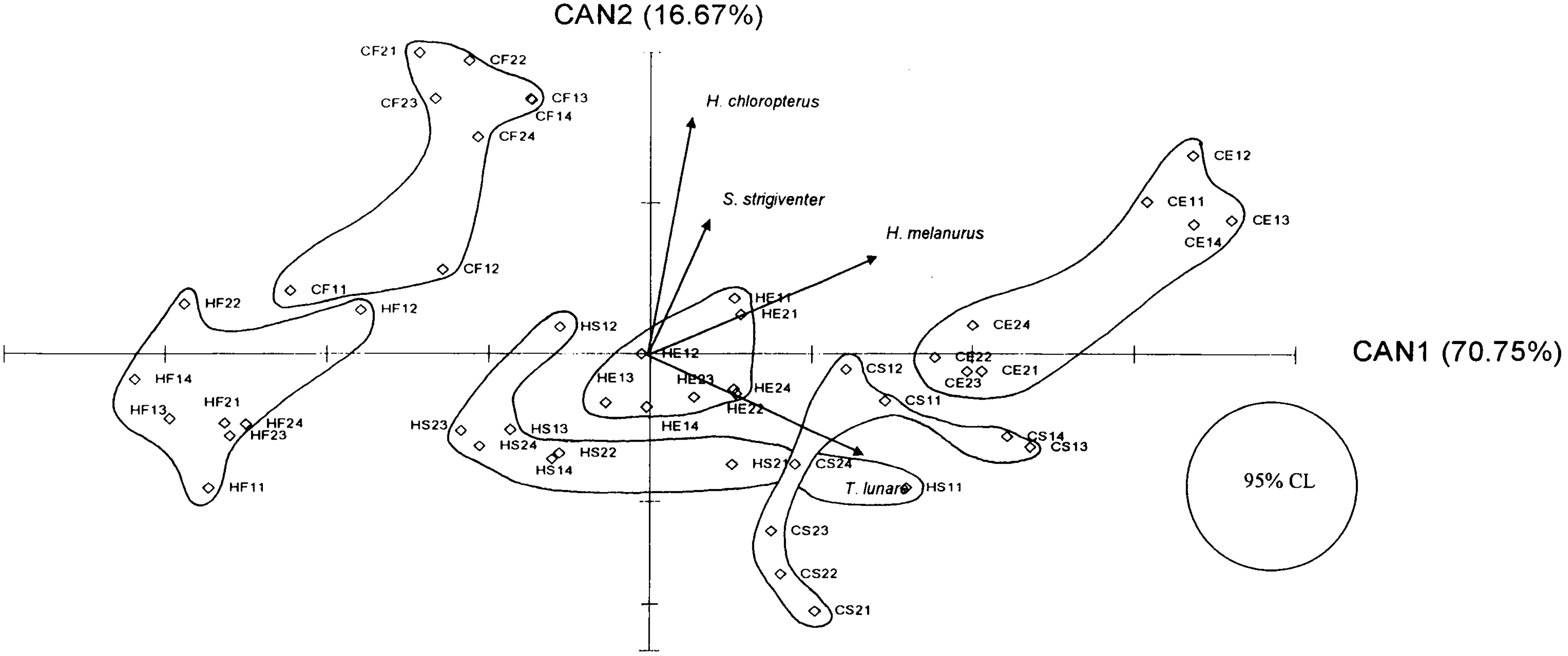


Fig. 3.2. Ordination plot from CDA showing the influence of eight different transect dimensions on abundance estimates of four wrasses at two localities (each with three habitats). The details of code names are as follows: the first letter indicates locality (C =Cattle Bay and H = Hazard Bay), the second letter indicates habitat (F = Reef flat, E = Reef edge, and S = Reef slope), the first number indicates transect width (1 = 2m and 2 = 5m), and the second number indicates transect length (1 = 30m, 2 = 50m, 3 = 70m, and 4 = 100m).

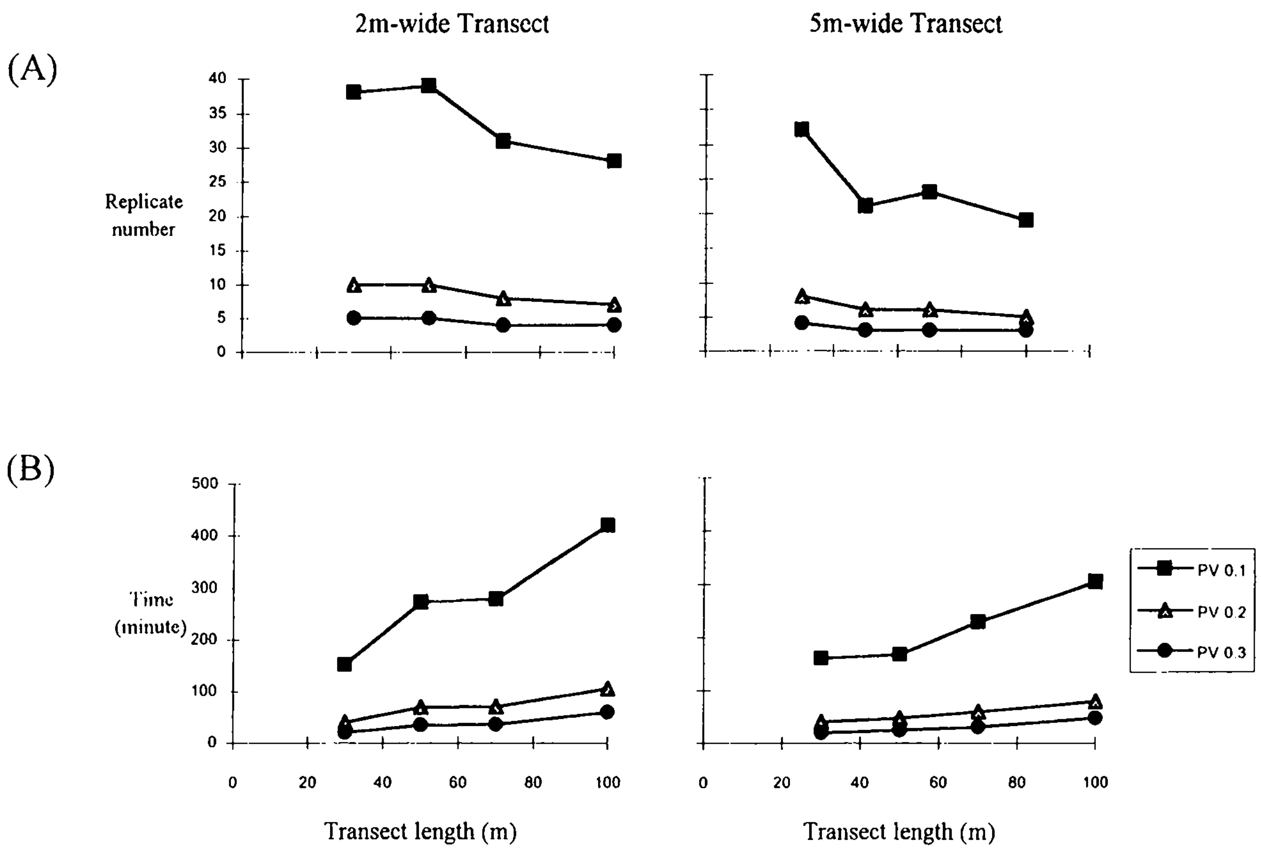


Fig 3.3. Optimised results: A) number of replicates, and B) sampling time, for two transect widths (2 and 5 m) and four transect lengths (30, 50, 70 and 100 m), at three precision levels (0.1, 0.2 and 0.3), averaged over four wrasse species: *Halichores melanurus*, *H. chloropterus*, *Thalassoma lunare* and *Stethojulis strigiventer*.

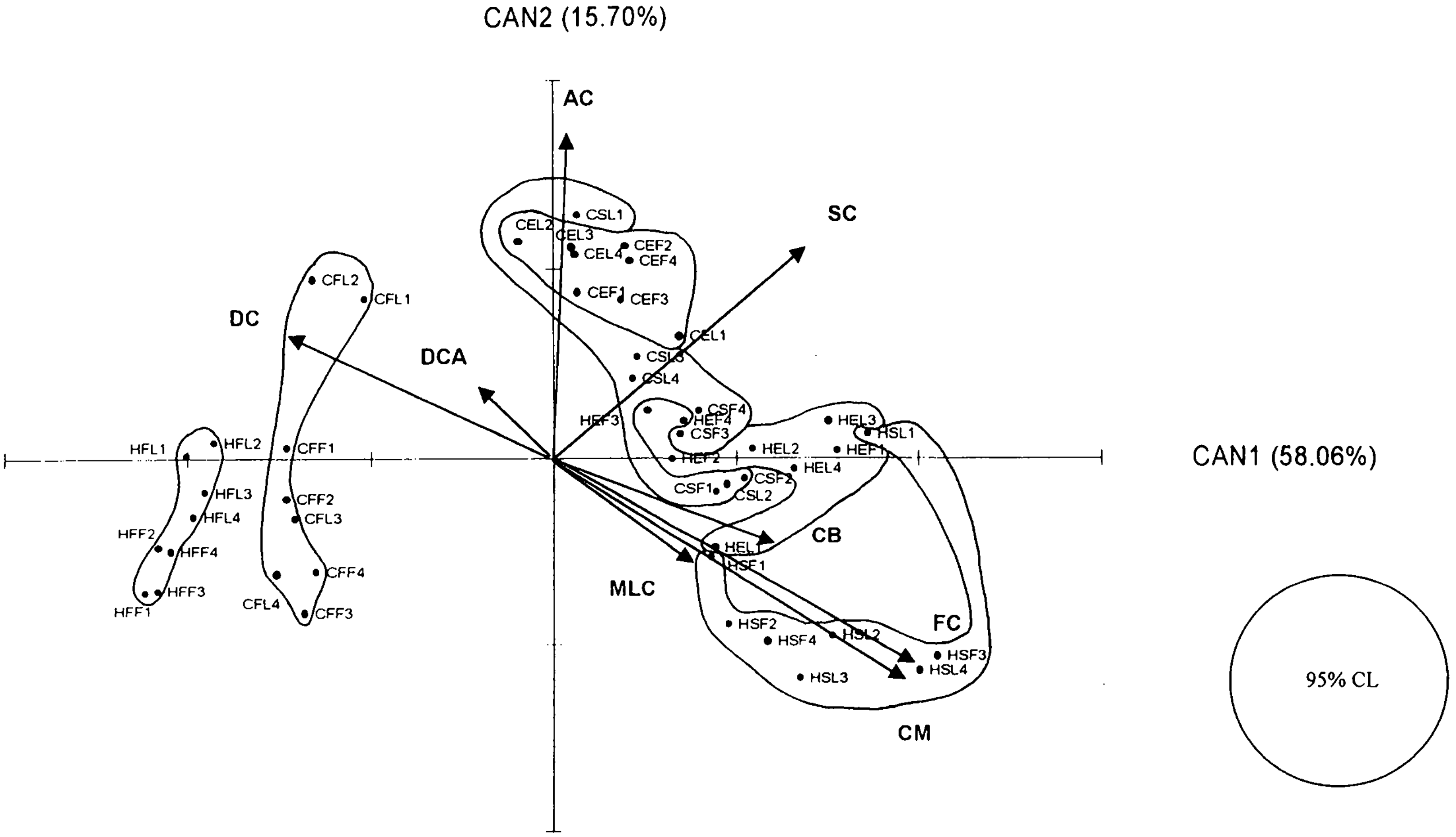


Fig. 3.4. Ordination plot from CDA showing the influence of two transect techniques and four transect lengths on % cover estimates of eight benthic categories over two localities and three habitats. The details of code name are as follows: the first letter indicates locality (C - Cattle Bay and H - Hazard Bay), second letter indicates habitat (F - Reef flat, E - Reef Edge, and S - Reef Slope), third letter indicates sampling technique (F - Fixed Density Point Transect and L - Line Intercept Transect) and the first number indicates transect length (1 - 30 m, 2 - 50 m, 3 - 70 m, and 4 - 100m). Reef benthic lifeform abbreviations are described in Table 3.1.

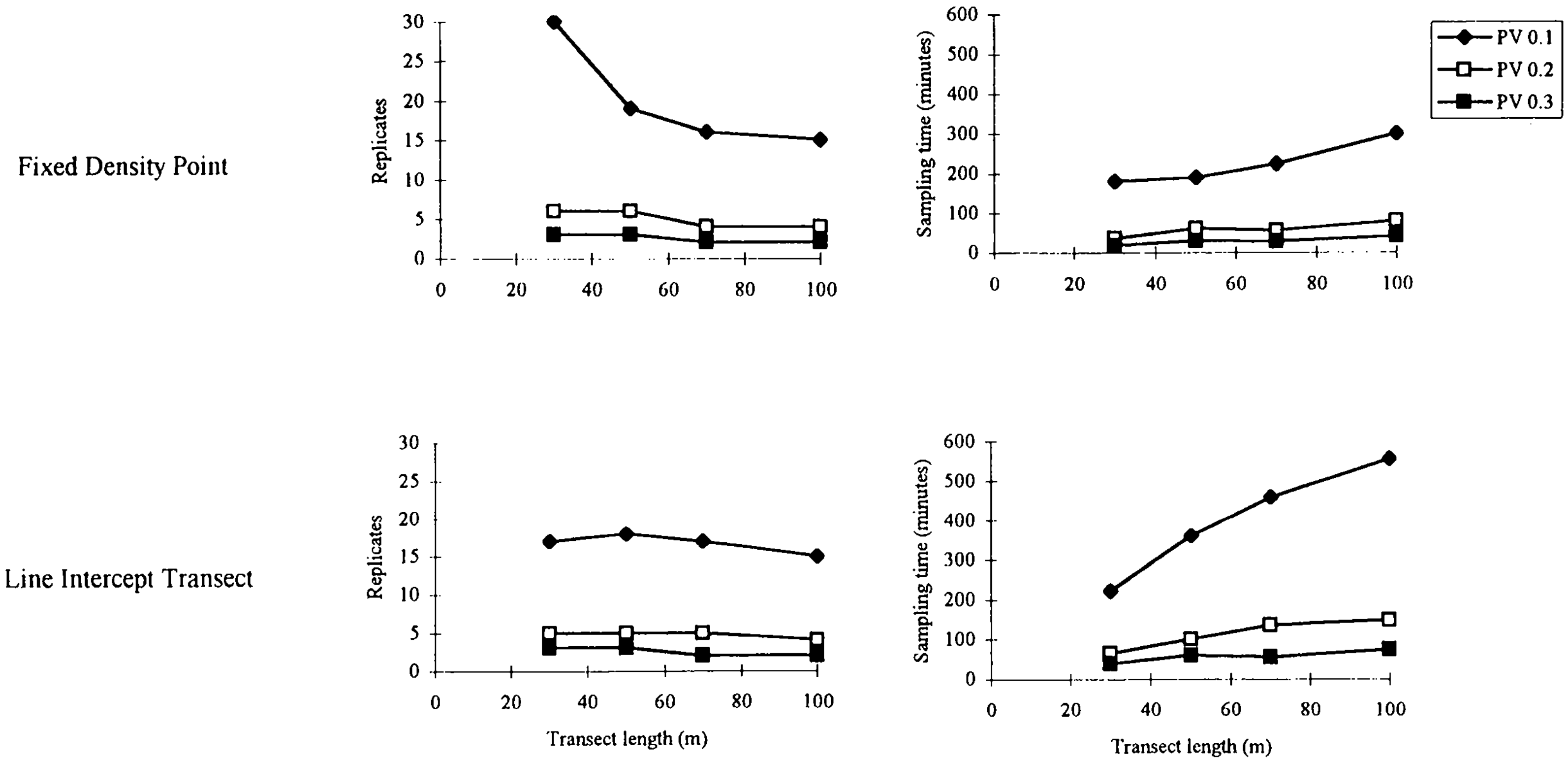


Fig. 3.5. Optimal number of replicates and sampling time for two sampling techniques (Fixed Density Point Transect and Line Intercept Transect) and four transect lengths (30, 50, 70, and 100m) at three Precision levels (0.1, 0.2, and 0.3) optimised for sampling eight reef benthic lifeforms (*Acropora* coral, Massive coral, Branching coral, Minor living coral component, Fire coral, Soft coral, Dead coral, and Dead coral with algae.

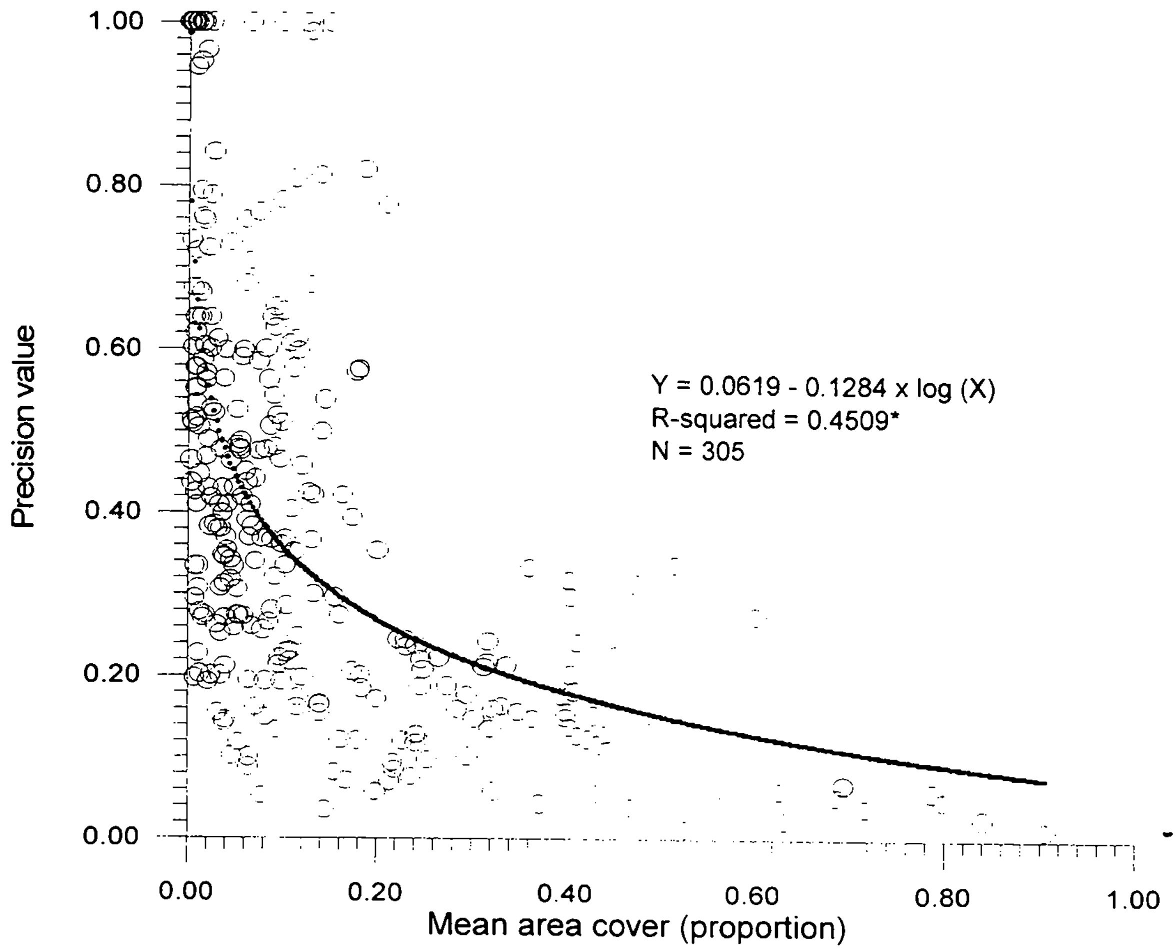


Fig. 3.6. Relationship between Mean area cover (X) and Precision value (Y) of reef benthos. Data were summarized for 10 benthic lifeforms sampled by two techniques over four transect lengths at two localities and three habitats: zero means were excluded from the calculation, * significant at P < 0.05.

# CHAPTER 4

# RELATIONSHIPS BETWEEN CORAL REEF FISH COMMUNITIES AND HABITAT STRUCTURE AT PHUKET ISLAND, THAILAND: A COMMUNITY-LEVEL ANALYSIS

## 4.1 SUMMARY

It has been suggested that a close relationship between the structure of coral reef fish assemblages and the coral habitat exists for only a few specialised groups of fishes. However, conclusions are often based on a few well studied locations, a few fish taxa and a limited array of methods. Here I investigated fish-habitat relationships at Phuket, Thailand during March 1994 and March 1995, by focusing on three important reef fish families: Labridae, Chaetodontidae and Pomacentridae. I used a comprehensive array of univariate and multivariate descriptors of fish-assemblage and biotic habitat-assemblage structure. Using univariate descriptors of fish communities, a variety of linear and parabolic relationships between living coral cover and community structure were observed by comparing multiple locations. Species richness was maximal at intermediate coral cover and evenness was maximal at the extremes of coral cover. Variation in the strength of these patterns determined overall patterns in species diversity. Temporal changes in fish communities were also partially influenced by changes in habitat structure, including an increase in coral cover at the reef slope of Kamala (recovery) and decrease at the reef edge of Lon, Hi, and Aoe (degradation) during the survey period.

Multivariate descriptors of fish communities and habitat variables indicated a strong association between the two. Canonical correlation analysis indicated that, in statistical terms, the Chaetodontidae were "responsive" to habitat structure (meaning that their presence/abundance are predictable by habitat structure), the Labridae were predictive (meaning that their presence/abundance are indicative of habitat structure), and the numerically dominant Pomacentridae were both responsive and predictive. All three families contained representative species that were associated with particular components of the biotic substratum. For example, branching and tabulate *Acropora* were closely associated with *Chaetodon triangulum, Labrichthys unilineatus, Hemiglyphidodon plagiometopon, Chromis viridis, Amblyglyphidodon leucogaster, and*

*Abudefduf sexfasciatus.* Canonical Discriminant Analyses carried out separately on the fish and habitat variables indicated extremely concordant patterns for labrid fishes and habitat structure compared with the other two taxa. The ordination based on habitat variables distinguished study sites on the basis of two broad regions, those west of Phuket and those south of Phuket (with the exception of Maiton Island). Between the two annual surveys, reef structure exhibited considerable changes at some sites, especially the reef slope of Niyang, Kamala, Patong, and Kata and the reef edge of Kamala, Lon, Hi, and Aoe. Greater changes in the species composition of fish assemblages were observed at these locations especially for pomacentrids which species diversity increasing during the survey.

Overall, results highlight a strong potential influence of the availability of different habitats on the abundance of species in three numerically important fish taxa in a relatively unstudied region threatened by habitat degradation.

## 4.2 INTRODUCTION

The degree to which the structure of coral reef fish assemblages is determined by the structure of coral reef habitats has been the subject of a number of investigations with many contrasting patterns described (Luckhurst & Luckhurst, 1978; Carpenter *et al.*, 1981; Bell & Galzin, 1984; Bell *et al.*, 1985; Bouchon-Navaro *et al.*, 1985; Roberts & Ormond, 1987; Roberts *et al.*, 1992; McClanahan, 1994; McCormick, 1994; Sale *et al.*,1984; Sale *et al.*, 1994). Much of this discussion centres on the degree to which the abundance of individual species or the diversity of different groups are correlated with the cover of live coral. It has been suggested that close relationships exist for only a limited number of families containing specialised species, such as the Chaetodontidae (Reese, 1981). This has led to the suggestion that this group may be superior as indicators of the condition of reef habitats (Reese, 1981). However, there are many reasons for differences in the strength of correlation with hard coral, including regional differences, differences in habitat types studied, the range of coral cover examined and the method of analysis (Sale, 1991a; McCormick, 1994; Jones & Syms, in press). In addition, consideration of only hard coral provides an incomplete picture of the importance of the coral reef habitat. Here I test the hypothesis that reef fish assemblages respond to spatial and temporal changes in the structure of the benthic habitat.

There have been two different approaches to investigating fish-habitat relationships at the community level. Many studies rely on standard univariate community indices such as species richness, species diversity, and evenness. These are subject to bivariate analyses of linear relationships between these measures and reduced measures of habitat structure such as % hard coral cover or topographic complexity (e.g. Luckhurst & Luckhurst, 1978; Carpenter *et al.*, 1981; Bell & Galzin, 1984; Bell *et al.*, 1985; Bouchon-Navaro *et al.*, 1985; Roberts & Ormond, 1987; McClanahan, 1994). Alternatively, studies have considered species-habitat relationships in terms of composition by employing multivariate statistical approaches (Dawson-Shepherd *et al.*, 1992; McCormick, 1994, 1995), which may be more sensitive in detecting responses to habitat (Dawson-Shepard *et al.*, 1992). Although it is not clear whether these two different approaches provide different conclusion or not, few workers have integrated these two approaches.

Studies of reef fish ecology have been concentrated in particular geographical areas, such as the Great Barrier Reef and the Caribbean (Sale, 1991a). A critical test of the ecological models that have been put forward is whether or not they apply to regions and assemblages with different biogeographical origins. Little is known about the ecology of reef fishes from the Andaman Sea in the north-east Indian Ocean, where coral reefs represent an important food source for coastal inhabitants. Like many other parts of the world, coral reefs in this region are threatened by various sources of disturbance, especially from human activities (Chansang *et al.*, 1981, 1992; Brown & Holley, 1982; Brown *et al.*, 1990; Chansang & Phongsuwan, 1993; Clarke *et al.*, 1993). At Phuket on the west coast of Thailand, reef monitoring programs were set up to examine the influence of coastal tin dredging (Chansang *et al.*, 1981; Chansang & Phongsuwan, 1993) and deep channel dredging (Brown *et al.*, 1990; Clarke *et al.*, 1993) on coral reefs. In these studies considerable effort was invested in monitoring changes in benthic components of the system. Although some studies on benthic assemblages and their associated reef fish assemblages were conducted at the same time (Chansang *et al.*, 1989), there have been few attempts to investigate the relationship between these two components.

The reefs of Phuket Island are located mainly along the west and south coastlines and around other nearby islands. On the west coast, reefs are open directly to clear oceanic water and are strongly influenced by the southwest monsoon. The physiomorphology of the reef in this area consists of an extensive reef flat and a reef slope extending to the depth of 15 m (Chansang & Phongsuwan, 1993). Offshore tin mining operations have been carried out nearby this reef since the late 19th century (Chansang, 1988) and substantial damage was first reported by Chansang *et al.* (1981). During the last ten years, however, tin deposits have been exhausted. A change in human use of the area has shifted the threats towards tourist-related activities and coastal recreation (Chansang & Phongsuwan, 1993). Some recovery of coral communities has already been reported for some of the degraded areas (Phongsuwan & Chansang, 1992).

Reefs of the south coast and near shore islands, in contrast, are found in shallow turbid waters of less than 10m depth. These areas are semi-protected from the southwest monsoon and are influenced by the turbid coastal waters of Phangna Bay. Reefs of this area were degraded by a combination of natural causes, such as unusual storms from the south (Phongsuwan & Chansang, 1992) and human activities, especially port construction and deep channel dredging (Brown *et al.*, 1990; Clarke *et al.*, 1993). The present status of this area is now uncertain because of the relocation of tin dredging from the depleted west coast and continued dredging activities. Knowledge of spatial and temporal patterns in habitat structure and responses of fish assemblages will be critical in evaluating these ongoing impacts.

The recent emphasis on a quantitative approach to problem solving has resulted in critical analyses of factors responsible for fish-habitat relationships. In this study, the nature of the relationship of coral reef fish assemblages and habitat structure was examined. Spatial and temporal patterns of habitat structure and the assemblages of three coral reef fish families (Chaetodontidae, Labridae, and Pomacentridae) were examined with respect to impact from human activities around Phuket Island.

## 4.3 MATERIALS AND METHODS

### 4.3.1 SAMPLING

This study was conducted along the west and south coasts of Phuket Island, Thailand (8°N, 98° 20′E). Spatial patterns in fish assemblages and habitats were assessed by surveying eight localities, four each from the west and south coasts of Phuket Island (Fig. 4.1): Niyang (A), Kamala (B), Patong (C), Kata (D), Lon Island (E), Hi Island (F), Aoe Island (G) and Maiton Island (H).

The fish assemblages and habitat structure within each locality were studied at two separate habitats, the reef edge and middle reef slope. The reef edge was 3 m below mean sea level (MSL) while the middle reef slope was 5 - 6 m below MSL. The term "study site" is used to refer to each habitat of each study locality.

Temporal changes to both the fish assemblage and habitats were assessed by sampling on two occasions, first during March 1994 and a year later during March 1995.

The community structure of three major fish families (Pomacentridae, Labridae and Chaetodontidae) was assessed using the visual census technique developed in Chapter 3. The census dimension used in this study was 30 m long and 5 m wide, with 6 replicates per transect. All fishes within a transect were identified to species level. The technique employed in this study was the "instantaneous" visual transect (Fowler, 1987) in which fishes were identified and counted as the transect line was laid down.

Habitat structure was assessed using the benthic life-form line intercept transect technique (LIT) (Loya, 1978; Dartnall & Jones, 1986). Based on the pilot study in Chapter 3, the transect length used in this part of the study was 30 m long and was conducted on the same transect line as the fish study after fishes were counted. The benthic substrata was recorded and classified into 20 lifeform categories (modified from Dartnall & Jones, 1986) which are considered to provide reasonable data quality in terms of precision (Chapter 3).

### 4.3.2 ANALYSIS

The raw benthic lifeform data were analysed and expressed as percentage area cover. A mean percentage area cover for each benthic category was calculated from six replicates. The abundance of all fishes at each habitat of each study site were calculated as mean abundance from the six replicates. Therefore, mean abundance of all fishes was expressed as individuals per 150 $m^2$. A Shannon-Wiener diversity index ($H' = -\Sigma p_i \log_e p_i$) and evenness index ($E = H'/H_{max}$) were calculated from mean abundance, using natural logarithms throughout (Pielou, 1974; Magurran, 1988). Habitat width ($AH'$) of each of the lifeforms/species were calculated according to the formula AH′ = 2.7183 H′ (Pielou, 1974) where H′ is the Shannon-Wiener diversity index calculated separately for each species across all sites.

The relationships between area cover of five major benthic lifeforms (total living coral, dead coral, algae, other fauna, and abiotics) and community parameters (total abundance, species richness, species diversity, and evenness index) for both habitat and fish assemblages were assessed. The relationships between community parameters of habitat structure with those of fish assemblages were also quantified. A series of linear and quadratic regression analyses were applied to describe the bivariate relationships. The relationships between parameters of each year were considered separately for 1994 and 1995.

*A priori* tests on both benthic lifeform and fish data was performed to satisfy the assumptions of parametric statistical methods, both univariate (Underwood, 1981) and multivariate (Bray & Maxwell, 1982). In most cases, a double square-root transformation provided the best results for habitat structure data. Therefore, the double square-root transformed ($x^{0.25}$) data was used for further parametric statistical analyses. This power transformation provides a comparable result with log (x) transformation (Field *et al.*, 1982 ; Clarke & Green, 1988). For fish data, the log (x+1) transformation provided the best result and was therefore selected. Both transformations were also used to reduce the chance that a few extremely dominant categories/species will dominate in further ordination analysis. For statistical hypothesis testing, a significance level of 0.05 was used throughout the study but the lowest level was also reported when appropriate.

A Principal Component Analysis (PCA) was performed to determine the influence of any family of fish that may dominate a whole fish assemblage, thus obscuring the influence of other families. PCA based on variance and covariance matrices were used to examine the pattern of the study sites derived from the variation of fish assemblages data. For fish, log (x+1) transformed data were used, which were centred but not standardized, because all of the data were on the same scale. An analysis was carried out with four sets of fish data, i.e., all three families of fishes combined and each family analysed separately. An analysis was executed using Proc PRINCOMP in SAS (SAS Institute Inc., 1990). Correlation between the results of all three families of fishes combined and each family were carried out on the principal component scores of the first three principal components using Spearman's rank correlation coefficients. Different correlation of each family with all three families data set, therefore, indicates different concordance result of each family with all three families.

Data reduction was needed as there were some rare variables present in the data matrix which would have the effect of reducing the power of further multivariate statistical analysis (Tabachnick & Fidell, 1989). Data reduction was carried out by using only reasonably common contributing variables or those with an occurrence in more than 20% of all sites (>6 sites).

A Multivariate Analysis of Variance (MANOVA) was used to test the hypothesis that key dependent variables were influenced by the factors in question. The Pillai's trace criterion was used in the MANOVA of this study because of its robustness over a wider range of conditions (Bray & Maxwell, 1982). The MANOVA model used was type I with all three fixed factors; time, locality and habitat. Times were treated as a fixed factor with two levels, March 1994 and March 1995 . Locality was treated as a fixed factor with 8 levels orthogonal within the time. Habitat was a fixed factor with two levels, reef slope and reef edge. An analysis was executed by Proc GLM in SAS (SAS Institute Inc., 1990).

Canonical Discriminant Analysis (CDA) was employed as *a posteriori* test for MANOVA to help identify the nature of any significant differences detected by MANOVA. CDA was performed on the centered log (x+1) data matrix and executed using Proc CANDISC in SAS (SAS Institute Inc., 1990). The canonical structure of each variable was used as "responsive" factor for the discrimination (i.e. the interpretation indicates variables would be most different for two centroids). Angular interpretation was thus used to interpret the ordination plot produced by CDA.

Canonical Correlation Analysis (CCA) was performed to determine the relationships between habitat structure and fish assemblages. CCA is analogous to running a multiple regression on the benthic and fish data separately, but constraining the generation of linear models so that they maximally correlate with one another (Tabachnick & Fidell, 1989). Double square root transformed data for habitat and log (x+1) transformed data for fishes were used in all analyses. Analyses was carried out for each fish family separately. The canonical intraset structure (correlation) were used to illustrated general trends of relationships between fish and habitat structure variables. Redundancy statistics was also carried out to show how much of the variation of the fish assemblage can be explained by the habitat structure variables, and vice-versa.

When variation of fish assemblage explained by habitat structure was greater than the variation of habitat structure explained by fish assemblage, these indicated the predictive role of habitat structure and conversely, the responsive role of the fish assemblage. Therefore, the presence/abundance of a predictive variable can indicate/predict opposite component variables and responsive role means that the presence/absence of these variables are predicted by opposite component variables. The analysis was performed using Proc CANCORR in SAS (SAS institute Inc., 1990).

## 4.4 RESULTS

### 4.4.1 UNIVARIATE COMMUNITY PARAMETERS

#### 4.4.1.1 Habitat structure

The habitat at the 16 sites at Phuket was dominated by dead coral, coral debris, massive corals, branching *Acropora*, foliose coral and dead coral with algae, both on the reef slope and reef edge. Together these habitats accounted for over 79.8% of the substratum but occurrence varied from 46.9% (dead coral with algae) to 100% (dead coral and massive coral) of the transects (Appendix 4.1). There was a temporal trend for each benthic lifeform especially on the reef slope where dead coral changes to coral debris and dead coral with algae (Fig. 4.2). Habitat width which indicates the relative distribution of each lifeform, suggested that high abundance benthic lifeforms usually have wide distribution.

Total % hard coral cover varied considerably among sites, zones and times (Fig. 4.3). Estimates varied between 20 and 60% on the reef slope, and 5 and 80% on the reef edge. Greatest cover was observed at Lon (E), Hi (F) and Aoe (G), and the lowest at Maiton Island (H). Cover was most often lowest at the reef edge, except for the three locations with greatest coral cover. These three sites also exhibited the greatest decline in coral cover between 1994 and 1995, with a decline of over 20% on the reef edge. Most sites changed very little. The only site that underwent an increase in coral cover was the reef slope at Kamala (B).

Comparison of the 16 sites showed that the category richness, diversity and evenness of benthic life-forms frequently exhibited a parabolic relationship with total hard coral cover and/or total dead coral, with greatest estimates observed at intermediate levels of cover, and low estimates at the extremes (Table 4.1). Areas of high coral cover tended to be dominated by a few dominant coral categories. These patterns were consistent between 1994 and 1995.

#### 4.4.1.2 Fish assemblages

The total of 101 fish species from the three families were recorded at Phuket during the 1994 and 1995 surveys, including 16 chaetodontids, 41 labrids and 44 pomacentrids (Appendix 4.2). Pomacentrids were numerically the most abundant fish families (ca. 85% of overall abundance), the most abundant species being *Neopomacentrus azysron, Pomacentrus moluccensis, Chromis viridis, Neopomacentrus anabatoides*, and *Plectroglyphidodon lacrymatus*. Labrids contributed less than 15% to overall abundance. The five most abundant labrids were *Thalassoma lunare, Labrichthys unilineatus, Halichoeres hortulanus, H. vrolikii,* and *Labroides dimidiatus.* Chaetodontids contributed less than 1% to the total fish abundance of the three families. The five most abundant species were *Chaetodon triangulum, C. collare, C. octofasciatus, C. trifasciatus,* and *Heniochus acuminatus.*

The number of species recorded varied among locations, but there was no consistent difference among the two reef zones (Fig. 4.4). The mean species richness on reef slope surveyed in 1995 was different from 1994 (Z = 2.11, P = 0.035. Wilcoxon paired-sample test) while those from reef edge surveyed in both year was not significantly different (Z = 1.47, P = 0.142) (Fig. 4.5A). On the reef slope, fish species richness at sites A, B, C, D, and F increased between 1994 and 1995 (Fig. 4.4). At both habitats, species diversity (Z = 2.11 and 0.98, P = 0.161 and 0.327) and evenness indices (Z = 0.42 and 0.84, P = 0.674 and 0.401) were not significantly different between years for both habitats, although there was a similar trend towards higher mean species diversity and evenness for both habitats in 1995 (Fig. 4.5B and 4.5C).

#### 4.4.1.3 Fish-habitat relationships

Patterns of fish species richness, diversity and evenness among sites were not correlated simply with the % total living coral excepted species diversity in 1994 (Spearman correlation coefficients, r = -0.5, P = 0.047). However, more complex, non-linear relationships between these parameters and either living or dead coral cover were apparent for some groups in 1994, when coral cover ranged from 5 to 80% cover (Table 4.1). Most of the detected relationships appeared to be polynomial (quadratic) with a few linear relationships also detected (Table 4.1). A parabolic relationship between species richness and % hard coral cover was detected for all species combined and labrids, with a maximum richness at intermediate levels of coral cover (40-60% cover) (Fig. 4.6). There was a tendency for evenness indices to exhibit the reverse pattern, with maximum evenness to community structure at the extremes of coral cover (Fig. 4.6). That is, where there are fewer fish species, the relative abundance of the species tends to be more similar. The combination of these two patterns produced somewhat unpredictable effects on diversity indices, although for labrids, there was a linear decline diversity with coral cover.

These patterns were mirrored by similar relationships with % dead coral cover, which tended to be stronger and explain more of the variation (Fig. 4.7). Both labrids and pomacentrids (and all species combined) exhibited a parabolic relationship between species richness and dead coral cover, with maximum species richness at intermediate levels. These groupings exhibited the reverse pattern for evenness. There was an absence of such relationships for 1995 when there was a much smaller range in coral cover (Table 4.1).

### 4.4.2 COMMUNITY STRUCTURE

#### 4.4.2.1 Habitat structure

A MANOVA on 18 common benthic life-forms showed significant effect of the second order interaction between time, locality and habitat (Table 4.2A). That is, the magnitude of change between 1994 and 1995 varied for different combinations of location and habitat. A CDA was used to visualise the spatio-temporal variation in habitat structure. The first two canonical axes accounted for 48% of the total variance and the ordination plot showed the relative change between 1994 and 1995 for the two different habitats at each location (Fig. 4.8). It was cleared that spatial variation in habitat structure among localities explained the greatest amount of variation in the data, contributing to both axes. Localities can be divided into two main arbitrary groups by axis 1; group 1 - A, C, D, and H, group 2 - B, E, F, and G. The benthic life-forms that dominated in group 1 were sand, fire coral, corallimorphs, massive coral and coral debris, while group 2 were characterised by branching *Acropora*, branching coral, foliose coral, tabulate *Acropora* and blue coral. Axis 2 identified another spatial trend causing heterogeneity in group 1. Location C was characterised by corallimorphs and massive corals, and locations A and D by sand and coral debris.

In general, differences among depth strata explained less variation, with no consistent trends among locations (Fig. 4.8). Habitat structure differed among depth strata at locations B, C, D and H , while other locations were more uniform. At location C, corallimorphs were found more on reef slope than on the reef edge. At H, sand was an important feature of the reef slope while coral debris as found on the reef edge.

Patterns of temporal variation were specific to locations and depth strata. Greatest changes in terms of the two main axes occurred at location C (reef slope and reef edge), the reef edge at locations E, F, G, H and the reef slope at A, B and D. At location C, corallimorphs increased from 1994 to 1995 in both reef habitats. At the reef slope of B, the submassive coral surveyed in 1995 was higher than in 1994. In contrast, branching *Acropora* at the reef edge of E, F and G surveyed decreased substantially between 1994 and 1995.

#### 4.4.2.2 Fish assemblages

The strong correlation between PCA scores for pomacentrids only and all three families combined (for the first three principle components) indicated that variation in the composition of the pomacentrid fauna dominated patterns at the whole community level (Table 4.3). MANOVA's and CDA's were, therefore, performed on each family separately in order to determine their individual patterns with respect to location, habitat zone and time . Eight species of chaetodontids and 24 species of both labrids and pomacentrids were analysed.

**Chaetodontidae**

The MANOVA carried out for chaetodontids only showed two significant first-order interactions among: 1) year and locality, and 2) locality and habitat (Table 4.2B). This result indicated that locality was the primary source of variation over year and habitat. For both labrids and pomacentrids, there were significant second-order interactions among the two years, eight localities and two habitats (Table 4.2C and 4.2D). The temporal patterns for habitat structure appeared to be site-specific.

CDA's were carried out to illustrate the nature of these interactions which indicate non-consistent results across all factors (Fig. 4.9). For chaetodontids, the first two canonical discriminants described 62% of the variation (Fig. 4.9A). The primary axis (42%) indicates a temporal trend specific to location F (both habitats) and the reef slope of C and E, where the abundance of *Heniochus acuminatus* and *Chaetodon* *trifasciatus* in decreased from 1994 to 1995, to be similar to the other locations. The second axis largely represents a spatial trend, with locations A, B and H characterised by *C. trifascialis* and *C. plebeius*, and locations D, F, and G by *C. triangulum* and *C. octofasciatus*. For these six locations, it appears that temporal variation was greater than differences among the two reef habitats. For example, fish on the reef slope and reef edge at locations A and G were distinct in the two years of sampling.

**Labridae**

The first two canonical discriminants for labrids explained 43% of the variation, and indicated consistent spatial differences that were more important than differences among habitats or years (Fig. 4.9B). It is clear that labrid assemblages at H (Maiton Is.) can be separated from the other localities, because the high abundance of *Halichoeres scapularis* and *Stethojulis interrupta*. Axis 1 detects a trend in community structure that tends to separate the localities along the west coast (A, B, C, and D) from those along the south coast (E, F, G), although there is also considerable temporal variation associated with this trend. Localities along the west coast have high abundance *of Halichoeres hortulanus, Stethojulis trilineata, Cheilinus trilobatus, Halichoeres marginatus, Thalassoma hardwicke*, and *Gomphosus caeruleus*, while localities along the south coast have high abundance *of Cheilinus fasciatus, Halichoeres vrolikii, Labrichthys unilineatus, Cheilinus digrammus, Bodianus mesothorax*, and *Epibulus insidiator*. Temporal variation in community structure was greater at some locations and/or habitats than others, with greatest change occurring at places where there was dramatic changes in either total coral cover or habitat structure. This was obvious for example, at the reef edge of H, where overall fish abundance in 1995 was lower than in 1994; in both habitats at F, where the abundance of the species characterising this area decreased between 1994 and 1995; and on the reef slope at B and D, where the abundance of characteristic species also decreased from 1994 to 1995; and on the reef slope at C, where the species composition showed greater affinities with the south coast group in 1994, but not in 1995.

**Pomacentridae**

For pomacentrids, the first two canonical discriminants explained 52% of the variation and illustrated the strong interaction between spatial and temporal variation (Fig 4.8C). Axis 1 mainly distinguished locations F and G from the other locations, with a high abundance of *Hemiglyphidodon plagiometopon, Chrysiptera rollandi, Amblyglyphidodon leucogaster, Chromis viridis, Neoglyphidodon nigrosis,* *Pomacentrus moluccensis* and *P. adelus* characteristic of F and G, and greater numbers of *Pomacentrus chrysurus* and *Pomacentrus similis* at other locations. Much of the temporal variation was in axis 2, indicating that some locations (particularly A and E) were characterised by high numbers of *Neopomacentrus anabatoides, Neoglyphidodon melas* and *Plectroglyphidodon lacrymatus* in 1994, while joining most other locations with a high abundance of *Neopomacentrus azysron* and *Chromis ternatensis* in 1995. Greatest temporal variation appeared to occur at locations A, E, F and G, the latter 3 being the locations recording the greatest decline in overall coral cover, particularly on the reef edge.

The comparison of the CDA results for the three fish families indicates considerable differences among the taxa. Of these, the labrids showed the strongest association with benthic characteristics detected by this analysis, although the pattern for pomacentrids was also strongly correlated with the first axis describing changes in habitat structure (Table 4.4).

#### 4.4.2.3 Fish-habitat relationships

The Canonical Correlation Analysis more explicitly addressed multivariate relationships between the structure of the three fish taxa and the structure of the habitat (Table 4.5A). The correlation were strongest and most similar for labrids and pomacentrids, while the chaetodontids showed weaker relationships. The results of redundancy analysis (the first three axis), showed that the amount of variation explained by their own variable set were moderate (27-54%) and those by opposite variable set were relatively small (10-29 %). The variance explained by opposite data sets indicated some differences among fish families in their response to or ability to predict to habitat structure (Table 4.5B). Chaetodontids appeared to be primarily responsive to habitat structure, rather than predictive (i.e. habitat structure explains fish structure better than vice versa). In contrast, labrids were more predictive of habitat structure (i.e. fish assemblage structure explains habitat structure better than vice versa), while pomacentrids were equally responsive and predictive. The variance explained by fish in the fish-habitat canonical correlation indicates that there is less variation at the transect level in chaetodontids (54%), compared with pomacentrids (38%) and labrids (31%) respectively.

The ordination plot from CCA of each fish family showed the trend in the relationships between fish species and particular benthic lifeforms (Fig. 4.10). Fish assemblage can be interpreted with respect to benthic lifeforms (or vice versa). For example, *Chaetodon triangulum* were most abundant where "Submassive Coral" (CS) and "Tabulate *Acropora*" (ACT) were most abundant (Fig. 4.10A); *Labrichthys unilineatus* was associated with "Branching *Acropora*" (ACB); *Halichoeres hortulanus, Stethojulis interrupta* and *Halichoeres scapularis* were associated with "Coral Debris" (CD) (Fig. 4.10B), and *Hemiglyphidodon plagiometopon, Chromis viridis, Amblyglyphidodon leucogaster* and *Abudefduf sexfasciatus* were associated most with "Branching *Acropora*" (ACB) and "Tabulate *Acropora*" (ACT) (Fig. 4.10C).

## 4.5 DISCUSSION

At the assemblage level, few studies have considered whether different fish assemblages show similar relationships with habitat across varied environments and whether environmental conditions affect different aspects of fish assemblages in a comparable manner (Jackson & Harvey, 1993). Bivariate and multivariate approaches provide different perspectives and different levels of information about these relationships, and together, provide the most comprehensive picture of covariation in fish and habitat parameters in space and time. Using these different approaches in this study for three common reef fish taxa (Chaetodontidae, Labridae and Pomacentridae), a variety of general and taxon-specific patterns emerged.

The bivariate approach, focusing on the relationships between either total living coral or dead coral and some community parameters for fishes, suggested that many of the relationships were nonlinear (parabolic) rather than linear. Similar parabolic relationships between community and habitat measures has been found in other coral reef studies (Grigg, 1983; McClanahan & Shafir, 1990). This relationship is usually a concave downward parabolic curve in which the intermediate value of population measure (e.g. % cover) has the highest value of community measure (e.g. species richness, diversity). At assemblage level, this relationship may be explained by the "intermediate disturbance" hypothesis (Connell, 1978), which predicts that communities will reach greatest species richness in moderately disturbed habitats. At extreme levels of disturbance, species richness is reduced through elimination of less tolerant species, and in benign environments that are seldom disturbed, species richness is reduced through competitive interactions. This model has been applied to coral communities in the past (Connell, 1978), and appears to be supported here, both for benthic habitat types and fish, both of which tended to exhibit greater richness at intermediate levels of total coral cover. However, whether competitive processes are involved or not is unknown.

There are a number of other potential explanations of this pattern. Intermediate levels of coral cover may represent an unstable condition, with very low and very high cover the more stable, undisturbed extremes. The stable, extreme environments may be characterised by a small number of competitively dominant species, while in the unstable environment, additional species with good colonising abilities may also be represented (see also Scarsbrook & Townsend, 1993; Townsend & Hildrew, 1994; Death, 1995).

The pattern would also be predicted, on the basis of patterns in habitat complexity, given a suite of species specialised on either live or dead coral substrata. Fishes present in habitats of high living coral cover should be only those species that have a close association with living coral. Likewise, fishes present in habitats of low living coral cover, dominated by dead coral or sand, should be those species that have a close association with dead coral or sand. When these extreme habitats are disturbed and are represented by both living coral and dead coral/sand, therefore, both groups of fish should be present, giving a higher overall species richness. This model is supported by studies which generally find a correlation between species richness and habitat complexity (e.g. Kohn, 1967; Roberts & Ormond 1987).

The concave downward parabola is also a characteristic of "Shelford's tolerance curve" (Putman, 1994) which appears to explain the relationship between living organisms and environmental parameters in some groups. The underlying principle is that any organism has a restricted range of environmental conditions and is thus limited by an upper and a lower lethal limit (beyond which the organism is ecologically inviable) (Putman, 1994). Within this limit is an optimum range in which the organism can maximise its activity.

At Phuket, there was only a "downward" parabola for the relationship between fish species richness and total living coral (or dead coral cover). The relationship between cover and evenness indices was a concave upward parabola. This additional result indicated that intermediate cover of living coral or dead coral not only provides habitat for more species of fish, but can lead to dominance by certain species. This dominance may relate to species that are specialised on habitats of intermediate coral cover, which may include "edge" associated species (Jones & Syms, *in press*).

The contrasting patterns in relation to species richness and evenness lead to few consistent relationships between species diversity indices and coral cover. The exception was labrids, which exhibited negative linear relationships with total living coral and positive relationships with dead coral. This suggested that there are more species or a greater variety of individuals associated with low coral cover in this group. Certainly, there are very few coral feeding or associated wrasses (e.g. *Chaetodon* spp., *Labrichthys unilineatus*), compared with small carnivores feeding over dead coral or sand areas (e.g. *Halichoeres hortulanus* and *Halichoeres scapularis*).

Multivariate descriptors of both fish and habitat variables indicated a strong association between the two, but the strength of these associations was dependent on the family of fishes in question. Locational differences were clearly the most important in both habitat and fish data sets. The CDA results indicated that coral reefs in the west differ in benthic composition to those south of Phuket, with Maiton Is. being an outliner. This appears to be primarily due to the influence of the SW-monsoon. The prominent feature of the benthic composition in the west Phuket (windward) was the abiotic components, dead coral with algae, sand and coral debris. The dominance of dead coral with algae and coral debris may indicate the effect of tin dredging in the past (Chansang *et al.*, 1985) and after the termination of this activity, the reef appears to have been further damaged by a monsoon. In contrast, living components especially *Acropora* branching corals, *Acropora* tabulate corals and foliose corals dominated in most sites south of Phuket (leeward), which may be exposed to more recent influences, such as sediment influx (Clarke *et al.*, 1993). Minor differences within each area were also detected, which might result from various activities whose type and effect varied among localities. For example, the reef at Maiton Island (H) differed from the rest of the south coast of Phuket. The reef at Maiton Island was dominated by coral debris and sand which indicates heavily degradation and no sign of recovery. Branching *Acropora* and foliose coral were the important features of the south Phuket reefs, which have been reported else where as having good tolerance to turbid water. Historically, the reef at Maiton Island was also dominated by branching *Acropora,* but has since been devastated by storm (Chansang, Phuket Marine biological Center, pers. comm.). The reef at Maiton Island lacks a wave barrier (coral) and coral debris is easily removed or buried by sand through wave action.

The labrids exhibited perhaps the closest relationship to the patterns in habitat structure. The labrids are one of the most speciose and abundant families of reef fish (Victor, 1986; McClanahan, 1994) and individual species exhibit wide patterns of distribution (Victor, 1986). Although their abundance and biomass is usually far below that of pomacentrids (Williams & Hatcher, 1983), their diversity is usually higher. In this study, labrids accounted only for 13% of overall abundance, while pomacentrids accounted for 85%, but both had similar species richness and diversity. They appear to be very responsive to habitat change and to be sensitive to low levels of disturbance. Their feeding mode varies from omnivores to carnivores (Sano *et al.*, 1984; 1987). Different groups in the family may respond to different phenomena on the reef. For example the coral feeding *Labrichthys unilineatus* may be present in habitats dominated by *Acropora* spp. which may be interpreted as a healthy reef. While *Halichoeres marginatus, H. vrolikii* and *Thalassoma lunare* are usually present in association with massive coral cover, which might be interpreted as a moderate reef condition. In addition, *Halichoeres hortulanus, H. timorensis, H. nigrescen, H. scapularis, Stethojulis interrupta* and *S. trilineata* are dominant in areas of coral debris or sandy bottoms which might be interpreted as degraded reef. This wide range of ecological modes may indicate why as a group, wrasses are reasonably good predictors of changes in habitat structure. In addition, their moderate abundance (McClanahan, 1994; Williams & Hatcher, 1983) when compared with other fish families may result in a relatively low census error, which may reduce unexplained variation in their abundance.

Since pomacentrids are mostly site attached and specialised species, they are usually considered to correlate highly with features of the habitat (Wellington & Victor, 1985; McCormick, 1994). In this study, the pattern in community structure did not provide the same close match with habitat that was observed for labrids (see CDA results). Nevertheless, the community was significantly correlated with the main trend in habitat structure (see CCA results). One factor contributing to the slightly poorer association with habitat variables (compared with labrids) may be that they are generalised to occupy a wider range of habitats. Most species found at Phuket have a broad habitat width (Appendix 4.2). Another factor is that their abundance is usually extremely high, which may make them slower to respond to any changes in their habitat (McClanahan, 1994). Their great abundance may also contribute to greater error in enumerating their abundance in the field, which will reduce the strength of any patterns detected.

The chaetodontid data was the weakest for identifying fish-habitat associations at Phuket. This may be because of their low diversity and abundance in this area, contributing less than 1% of overall fish abundance. Chaetodontids have been reported to respond to acute effect habitat changes (Reese, 1981; Sano *et al.*, 1984, 1987; Bouchon-Navaro *et al.*, 1985; Williams, 1986). In view of the ease if visual identification of species in this group, they have therefore, been suggested as good indicators of habitat degradation (e.g. Reese, 1981; Sano *et al.*, 1984, 1987; Roberts *et al.*, 1987). However, this study suggests that they do not have the best biological characteristics for that purpose in this area.

Considerable differences in fish assemblages in all three families were detected among years, and some of this variation was related to changes in habitat structure. Locations exhibiting large changes in coral cover or habitat structure tended to exhibit parallel changes in fish communities, but the details of the responses were specific to each family. For example, substantial temporal changes in the species composition of pomacentrids and an overall increase in fish species diversity occurred at locations (E, F and G) that underwent a major decline in coral cover (from 80 to 50%). Other changes were less clearly explained by habitat structure. For example, labrid and chaetodontid species richness and species diversity did not follow the same trend as pomacentrids.

The most striking results from total living coral cover were from the fast recovery at the reef slope of site B and the degradation of the reef edge of localities along the south coast (E, F, G; H being exceptional in that it was already destroyed). These contrasting results were apparently caused by a gradient of human impact from west to south. On the west coast (A, B, C, D), the coral assemblage had been degraded in the past by tin mining, but this has ended allowing corals to recover. Recovery of living coral in this area has already been observed (Phongsuwan & Chansang, 1992). In general, live coral at several sites on the west coast was increasing, but only at site B on the slope was this significant. For the south coast of Phuket, the systematic degradation on the reef edge indicated that the cause of degradation was depth related. It appears during the survey in 1994 that a minor oil spill usually occurred in this area (personal observation) which may have caused the death of living coral during low tide. The sources of the oil spill was not known but it is likely to have come from ships using the Phuket Deep Sea Port. Effects of oil spill on corals have been reported elsewhere (Loya & Rinkevich, 1980; Guzman *et al.*, 1994). Changes of major coral categories at the reef edge of E, F, and G between 1994 and 1995 also varied among sites (Table 4.6) which indicates site specific recovery? after systematic impact. At sites E and F, foliose coral (CF) initially dominated the reef edge, but in 1995 it was degraded to dead coral (DC) at site E and dead coral with algae (DCA) at site F in 1995. Site F had intense tourism activities which may have been a source of enhanced nutrients for algae growth. *Acropora* coral dominated at site G but it was degraded in 1995 to coral debris (dead coral fragments) which may also associated with strong wave action after the corals died. Sites E and G were only subjected to use by local fishermen fishing by hand lines, which is unlikely to impact corals. Reef monitoring of these localities is needed with more frequent surveys per year to determine the likely impact source.

Sano *et al.* (1987) reported the long-term effects of destruction of hermatypic corals on reef fish communities at Ireomote Island, Japan. They found that the number of fish species and the abundance of resident species (mostly pomacentrids) markedly decreased on the dead reef and rubble reef but the species numbers of the transient species (including labrids and chaetodontids) did not differ between living reef and dead reef, but decreased on rubble reefs. Sano *et al.* (1987) suggested that the factors responsible for these decreases were living space and food. Walsh (1983), in contrast, reported no decreases in number of species and abundance after habitat degradation by a storm. These conflicting results are likely to depend on the degree of disturbance. Sano *et al.* (1987) studied a collapsed system, while Walsh (1983) did not. McClanahan (1994) concluded that the effect from fishing on labrids and pomacentrids is moderate. McClanahan (1994) also found that the species richness of labrids and diodontids increased in areas unprotected fishing, while most families decrease. If habitats are destroyed, most of the site attached or coral-dependent species should disappear in a short time and might not be able to recover unless the habitat recovers. Therefore, there was more space for other species to occupy, especially species that are not dependent on corals. Most labrids have a high potential to occupy this available space. This response can be detected at any temporal scale, which indicates their sensitivity to a wide range of reef conditions. Recruitment of fish is also significant for recovery of reef fish assemblage which many reef fish species showed habitat specific for settlement on the reef (Sale, 1991b). Patterns of recruitment for reef fishes, however, are usually vary with both space and time (Doherty, 1991).

The fish-habitat relationships can change not only because time, but the sampling design as well. The intrinsic tendency for temporal change may differ for fish and corals. Some aspects of fish assemblages may change on a daily scale (Williams, 1983b), while habitat structure may remain static, even on an annual scale, without a major impact (Moran *et al.*, 1991). Conversely, when habitat structure does suddenly change (e.g. due to storm devastation), fish assemblages might be buffered through their ability to temporarily migrate (e.g. Walsh, 1983). Temporal scale, therefore, should be considered more carefully for the study of fish-habitat relationships. When sampling a limited number of sites, observed patterns could be unduly influenced by a single site. However, if the number of study sites is high, this problem should not occur.

Species richness, species diversity and evenness indices are useful ways of displaying the quality of a community, so these parameters have remained a central theme in ecology (Magurran, 1988; Putman, 1994). Studies on reef fish and habitat relationships at the disturbed reef of Phuket, however, showed that these parameters of reef fish did not always respond to changes in the quality of habitat, when measured in simple terms, such as total live or dead coral cover. Multivariate procedures revealed much more information about the specific effects of particular aspects of habitat structure on fishes and highlighted many differences in family-level responses. Coral reef fish assemblages at Phuket were dominated by the pomacentrids, with this family contributing most to overall patterns to the reef fish community. The greater robustness of this family to environmental change may mask the dynamic changes those are occurring in families that are more sensitive to habitat characteristics, such as the labrids.

The study indicated that reefs around Phuket are in different stages of recovery and degradation. Looking at the relationship between habitat structure and fish assemblages in a number of different families over longer time-scales will provide the most complete picture of the range of fish-habitat relationships and the impacts that environmental change has on reef fish assemblages.

Table 4.1. Temporal variation of relationships between living coral and dead coral with community parameters of some reef fishes and benthic assemblages When neither linear or quadratic models are significance, results from linear regression are presented. * significance at $P < 0.05$.

| Community parameters | Living coral | | | | | | | | Dead coral | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | 1994 | | | | 1995 | | | | 1994 | | | | 1995 | | | |
| | Model | P | $r^2$ | P | Model | P | $r^2$ | P | Model | P | $r^2$ | P | Model | P | $r^2$ | P |
| Overall richness | **$y = 24.96 + 62.77x - 74.69x^2$** | **<0.05*** | **0.35** | **0.06** | $y = 34.20 + 10.16x$ | 0.230 | 0.10 | 0.230 | **$y = 10.51 + 114.97x - 115.55x^2$** | **<0.05*** | **0.45** | **<0.05*** | $y = 39.18 - 2.27x$ | 0.791 | 0.005 | 0.791 |
| Overall diversity | $y = 2.36 - 0.48x$ | 0.183 | 0.12 | 0.183 | $y = 2.55 - 0.54x$ | 0.279 | 0.08 | 0.279 | $y = 1.87 + 0.59x$ | 0.136 | 0.15 | 0.136 | $y = 2.13 + 0.52x$ | 0.289 | 0.08 | 0.289 |
| Overall eveness | **$y = 0.84 - 1.24x + 1.27x^2$** | **<0.05*** | **0.36** | **0.054** | $y = 0.72 - 0.20x$ | 0.119 | 0.17 | 0.119 | **$y = 0.91 - 1.67x + 1.85x^2$** | **<0.05*** | **0.46** | **<0.05*** | $y = 0.58 + 0.15x$ | 0.219 | 0.11 | 0.219 |
| Labridae richness | **$y = 15.14 + 24.28x - 34.52x^2$** | **<0.05*** | **0.72** | **<0.05*** | $y = 16.44 + 0.77x$ | 0.836 | 0.003 | 0.836 | $y = 16.58 + 87.78x$ | 0.169 | 0.13 | 0.169 | $y = 16.06 + 1.81x$ | 0.623 | 0.02 | 0.623 |
| Labridae diversity | **$y = 2.21 - 0.73x$** | **<0.05*** | **0.46** | **<0.05*** | $y = 1.89 - 0.12x$ | 0.807 | 0.004 | 0.807 | **$y = 1.52 + 0.78x$** | **<0.05*** | **0.42** | **<0.05*** | $y = 1.74 + 0.28x$ | 0.550 | 0.03 | 0.550 |
| Labridae eveness | $y = 0.74 - 0.16x$ | 0.081 | 0.20 | 0.081 | $y = 0.68 - 0.06x$ | 0.748 | 0.008 | 0.748 | **$y = 0.81 - 0.99x + 1.19x^2$** | **<0.05*** | **0.43** | **<0.05*** | $y = 0.63 + 0.70x$ | 0.693 | 0.01 | 0.693 |
| Pomacentridae richness | $y = 11.39 + 1.41x$ | 0.614 | 0.02 | 0.614 | $y = 13.05 + 8.38x$ | 0.131 | 0.16 | 0.131 | **$y = 5.89 + 34.78x - 38.63x^2$** | **<0.05*** | **0.33** | **<0.05*** | $y = 16.75 - 0.82x$ | 0.886 | 0.001 | 0.886 |
| Pomacentridae diversity | $y = 1.60 - 0.08x$ | 0.823 | 0.004 | 0.823 | $y = 1.78 + 0.17x$ | 0.731 | 0.009 | 0.731 | $y = 1.49 + 0.16x$ | 0.701 | 0.01 | 0.701 | $y = 1.75 + 0.25x$ | 0.601 | 0.02 | 0.001 |
| Pomacentridae eveness | $y = 0.66 - 0.04x$ | 0.777 | 0.006 | 0.777 | $y = 0.69 - 0.07x$ | 0.610 | 0.02 | 0.610 | **$y = 1.06 - 2.26x + 2.42x^2$** | **<0.05*** | **0.35** | **0.060** | $y = 0.63 + 0.09x$ | 0.456 | 0.04 | 0.456 |
| Lifeforms richness | **$y = 8.91 + 23.19 - 28.08x^2$** | **<0.05*** | **0.37** | **<0.05*** | $y = 10.05 + 4.67x$ | 0.113 | 0.17 | 0.113 | $y = 11.58 + 1.15x$ | 0.666 | 0.01 | 0.666 | $y = 13.07 - 2.98x$ | 0.320 | 0.07 | 0.320 |
| Lifeforms diversity | $y = 1.37 + 0.34x$ | 0.307 | 0.07 | 0.307 | **$y = 0.67 + 4.86x - 5.08x^2$** | **<0.05*** | **0.62** | **<0.05*** | **$y = 0.86 + 4.18x - 4.92x^2$** | **<0.05*** | **0.49** | **<0.05*** | **$y = 1.27 + 3.46x - 5.27x^2$** | **<0.05*** | **0.64** | **<0.05*** |
| Lifeforms eveness | $y = 0.54 + 0.18x$ | 0.142 | 0.15 | 0.142 | **$y = 0.34 + 1.68x - 1.79x^2$** | **<0.05*** | **0.52** | **<0.05*** | **$y = 0.41 + 1.41x - 1.72x^2$** | **<0.05*** | **0.55** | **<0.05*** | **$y = 0.53 + 1.23x - 1.87x^2$** | **<0.05*** | **0.61** | **<0.05*** |

Table 4.2. Results of MANOVA showing spatio-temporal variation of habitat structure and fish assemblages: A) Habitat structure, B) Chaetodontidae, C) Labridae, and D) Pomacentridae, at eight localities each with two habitats from Phuket (* = significant at < 0.05, $^{ns}$ = non-significant).

| A) Habitat structure | df | F | P |
|---|---|---|---|
| Time, A | 18, 143 | 10.38 | 0.0001* |
| Locality, B | 126, 1043 | 15.92 | 0.0001* |
| AxB | 126, 1043 | 2.50 | 0.0001* |
| Habitat, C | 18, 143 | 11.47 | 0.0001* |
| AxC | 18, 143 | 1.23 | 0.248$^{ns}$ |
| BxC | 126, 1043 | 3.18 | 0.0001* |
| AxBxC | 126, 1043 | 1.63 | 0.0001* |

| B) Chaetodontidae | df | F | P |
|---|---|---|---|
| Time, A | 8, 153 | 9.99 | 0.0001* |
| Locality, B | 25, 1113 | 5.38 | 0.0001* |
| AxB | 56, 1113 | 3.09 | 0.0001* |
| Habitat, C | 8, 153 | 2.87 | 0.005* |
| AxC | 8, 153 | 1.50 | 0.163$^{ns}$ |
| BxC | 56, 1113 | 1.90 | 0.0001* |
| AxBxC | 56, 1113 | 1.22 | 0.132$^{ns}$ |

| C) Labridae | df | F | P |
|---|---|---|---|
| Time, A | 24, 137 | 3.47 | 0.0001* |
| Locality, B | 168, 1001 | 8.45 | 0.0001* |
| AxB | 168, 1001 | 2.74 | 0.0001* |
| Habitat, C | 24, 137 | 12.42 | 0.0001* |
| AxC | 24, 137 | 2.21 | 0.0024* |
| BxC | 168, 1001 | 3.43 | 0.0001* |
| AxBxC | 168, 1001 | 2.13 | 0.0001* |

| D) Pomacentridae | df | F | P |
|---|---|---|---|
| Time, A | 24, 137 | 25.39 | 0.0001* |
| Locality, B | 168, 1001 | 11.41 | 0.0001* |
| AxB | 168, 1001 | 7.22 | 0.0001* |
| Habitat, C | 24, 137 | 20.64 | 0.0001* |
| AxC | 24, 137 | 8.98 | 0.0001* |
| BxC | 168, 1001 | 8.66 | 0.0001* |
| AxBxC | 168, 1001 | 4.01 | 0.0001* |

Table 4.3. Spearman's correlation coefficients of the first three principal components for observation scores showing the influence of major coral reef fish families on the overall three family assemblages (* significant at $P < 0.05$).

| All three families | PC 1 | PC 2 | PC 3 |
|---|---|---|---|
| Chaetodontidae | 0.466* | 0.140 | 0.106 |
| Labridae | 0.750* | -0.183 | -0.405 |
| Pomacentridae | 0.994* | 0.996* | 0.982* |

Table 4.4. Spearman's correlation coefficients of the first three canonical discriminants for site scores showing different degree of concordance CDA results between habitat structure and three families of coral reef fish (* significant at $P < 0.05$).

| Habitat structure | CD 1 | CD 2 | CD3 |
|---|---|---|---|
| Chaetodontidae | 0.025 | 0.545* | 0.218 |
| Labridae | 0.526* | 0.386* | 0.650* |
| Pomacentridae | 0.464* | -0.188 | -0.132 |

Table 4.5. Results of canonical correlation analysis and redundancy analysis (the first three axes) showing the relationships between coral reef fish assemblages and habitat structure.

| | Chaetodontidae | | Labridae | | Pomacentridae | |
|---|---|---|---|---|---|---|
| A) Canonical Correlation analysis | | | | | | |
| | Correlation | % variance | Correlation | % variance | Correlation | % variance |
| Axis 1 | 0.67 | 38.5 | 0.87 | 31.0 | 0.85 | 26.0 |
| Axis 2 | 0.55 | 19.7 | 0.79 | 17.0 | 0.83 | 21.1 |
| Axis 3 | 0.49 | 14.9 | 0.76 | 13.9 | 0.73 | 11.3 |
| B) Redundancy statistics (% cumulative variance explained) | | | | | | |
| | Habitat | Fish | Habitat | Fish | Habitat | Fish |
| Explained by their own variable set | | | | | | |
| Axis 1 | 14.1 | 21.7 | 20.2 | 14.2 | 11.2 | 10.0 |
| Axis 2 | 22.2 | 44.5 | 34.3 | 22.6 | 30.2 | 24.5 |
| Axis 3 | 27.1 | 53.6 | 42.6 | 30.1 | 38.6 | 37.8 |
| Explained by opposite variable set | | | | | | |
| Axis 1 | 6.4 | 9.9 | 15.2 | 11.2 | 8.2 | 7.3 |
| Axis 2 | 8.8 | 16.7 | 23.9 | 16.0 | 21.2 | 17.2 |
| Axis 3 | 10.0 | 18.9 | 28.7 | 20.6 | 25.7 | 24.4 |

Table 4.6. Summarised of the major benthic area cover (%) at Lon Island (E), Hi Island (F), and Aoe Island (G) during 1994 and 1995.

| Benthic Categories | Lon Island | | Hi Island | | Aoe Island | |
|---|---|---|---|---|---|---|
| | 1994 | 1995 | 1994 | 1995 | 1994 | 1995 |
| Dead coral | 5.11 | 32.39 | 19.58 | 18.89 | 21.19 | 15.78 |
| Dead coral with algae | 0 | 0.06 | 0.33 | 22.03 | 0 | 5.83 |
| Coral debris | 10.28 | 9.42 | 1 | 0 | 0 | 24.61 |
| Foliose coral | 66.08 | 43.97 | 31.11 | 21.11 | 1.86 | 1.44 |
| Acropora coral | 2.83 | 7.19 | 31.94 | 26.33 | 66.36 | 41.69 |

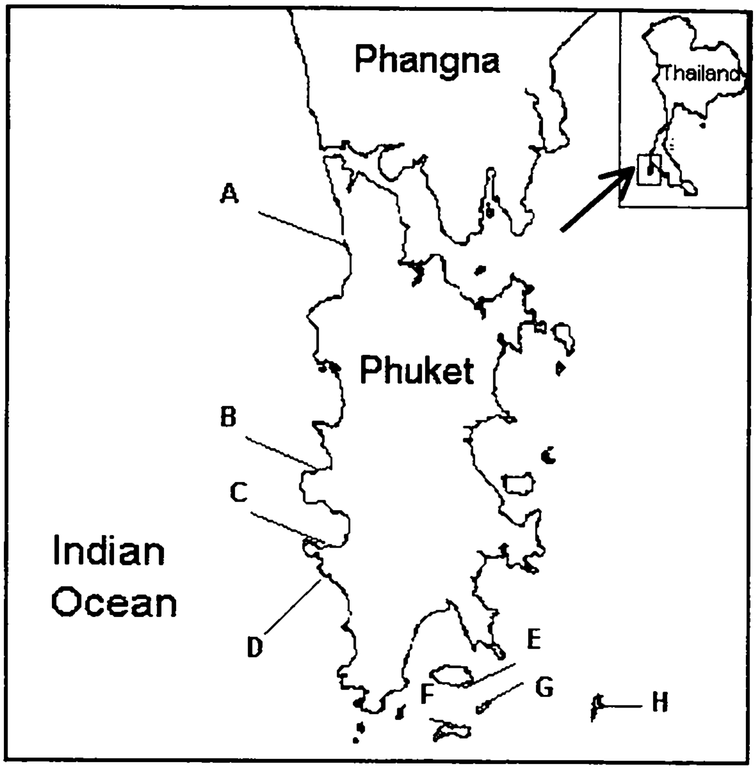


Fig. 4.1. Map showing study localities at Phuket, Thailand. A - Niyang, B - Kamala, C - Patong, D - Kata, E - Lon Island, F - Hi Island, G - Aoe Island, and H - Maiton Island.

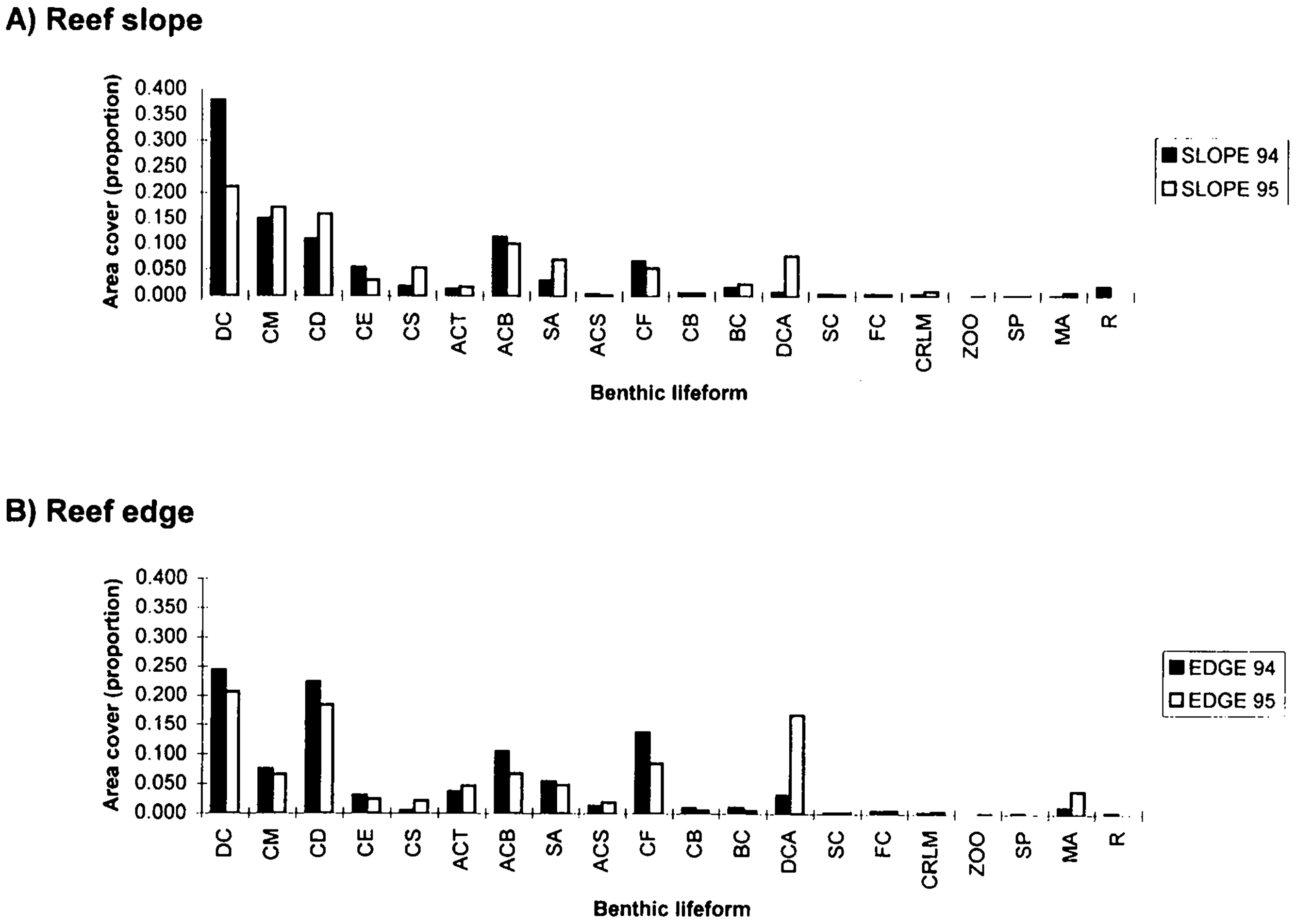


Fig. 4.2. Graphs showing temporal variation of benthic lifeforms (averaged across eight localities at A) slope and B) edge habitats in reefs at Phuket, Thailand surveyed during 1994 and 1995. Benthic lifeforms are as defined in Appendix 4.1

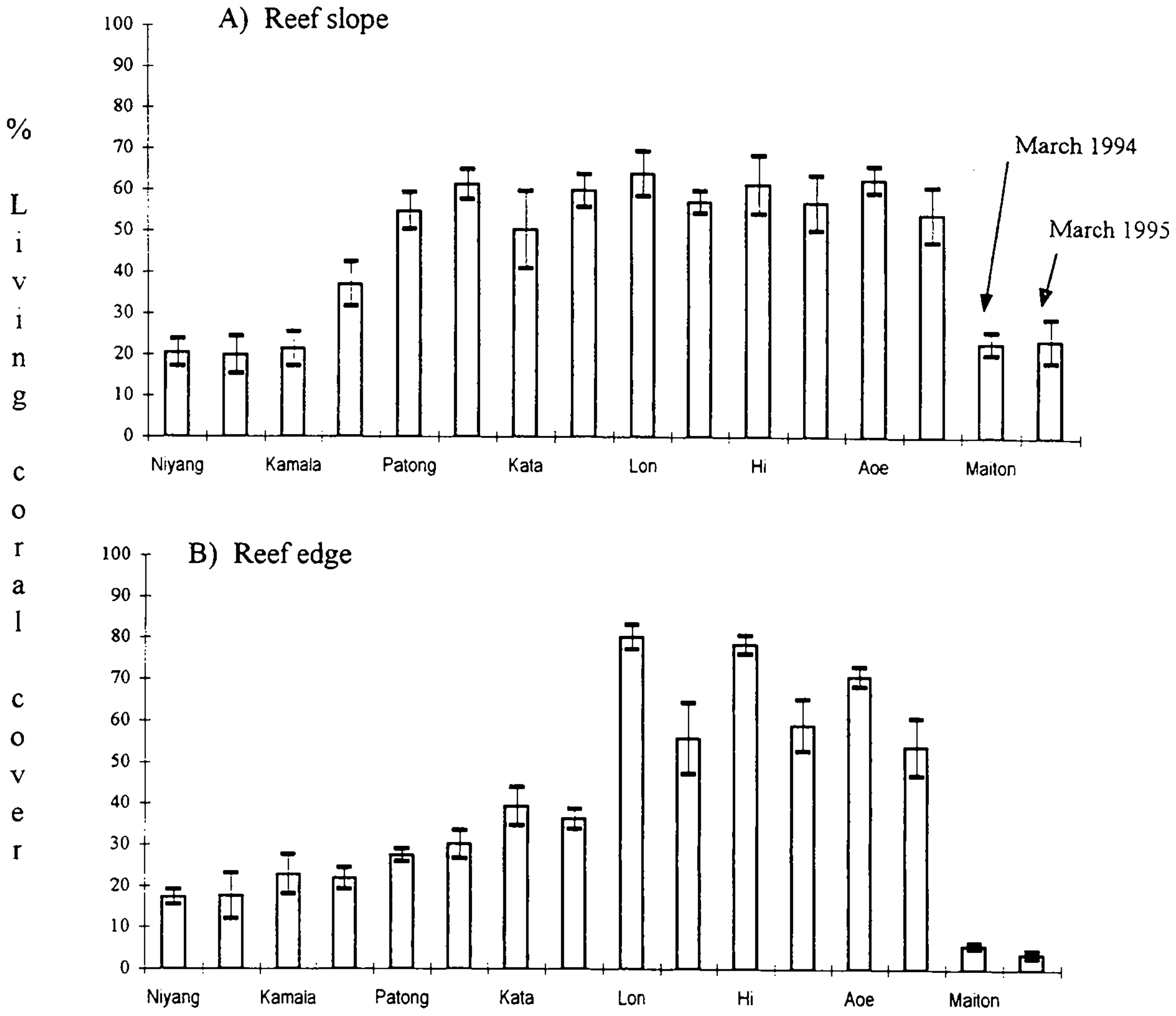


Fig. 4.3. Mean total area cover of living coral ( ± SE) at two habitat: A) Reef slope and B) Reef edge, at eight localities of Phuket surveyed in 1994 and 1995.

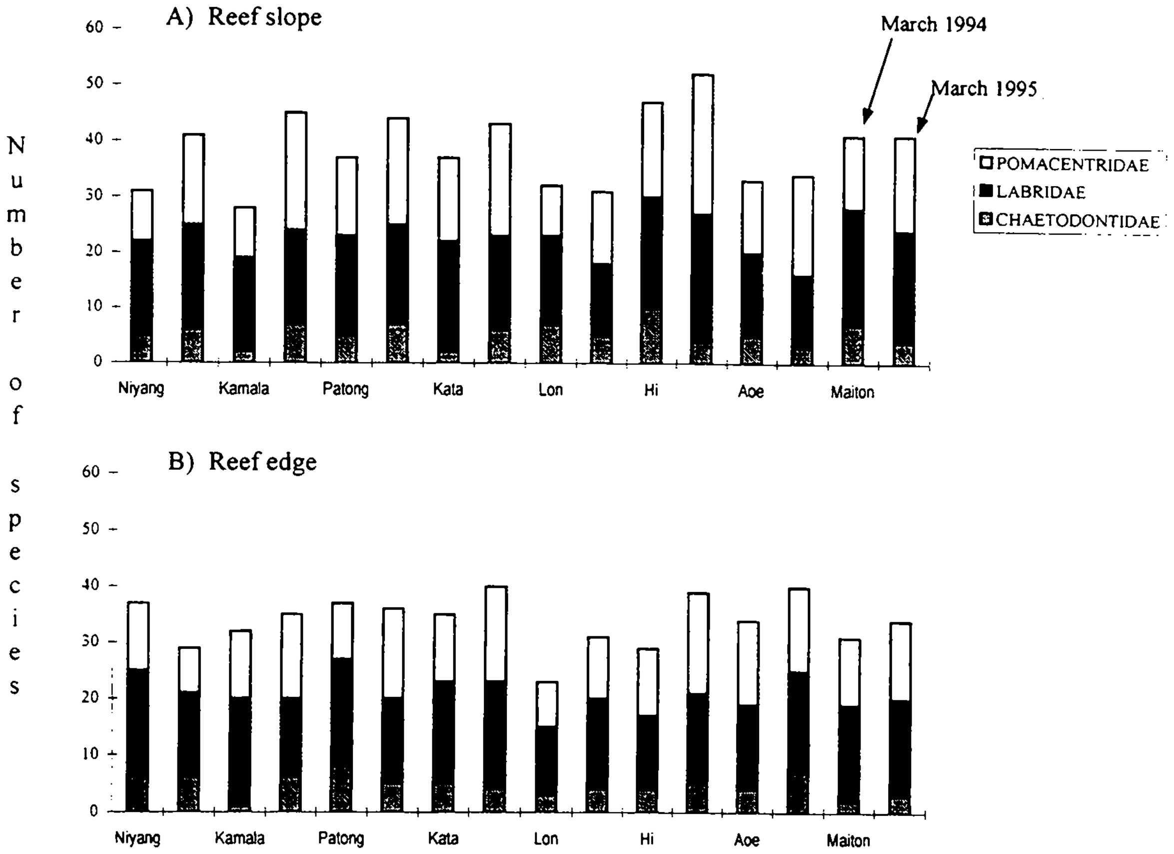
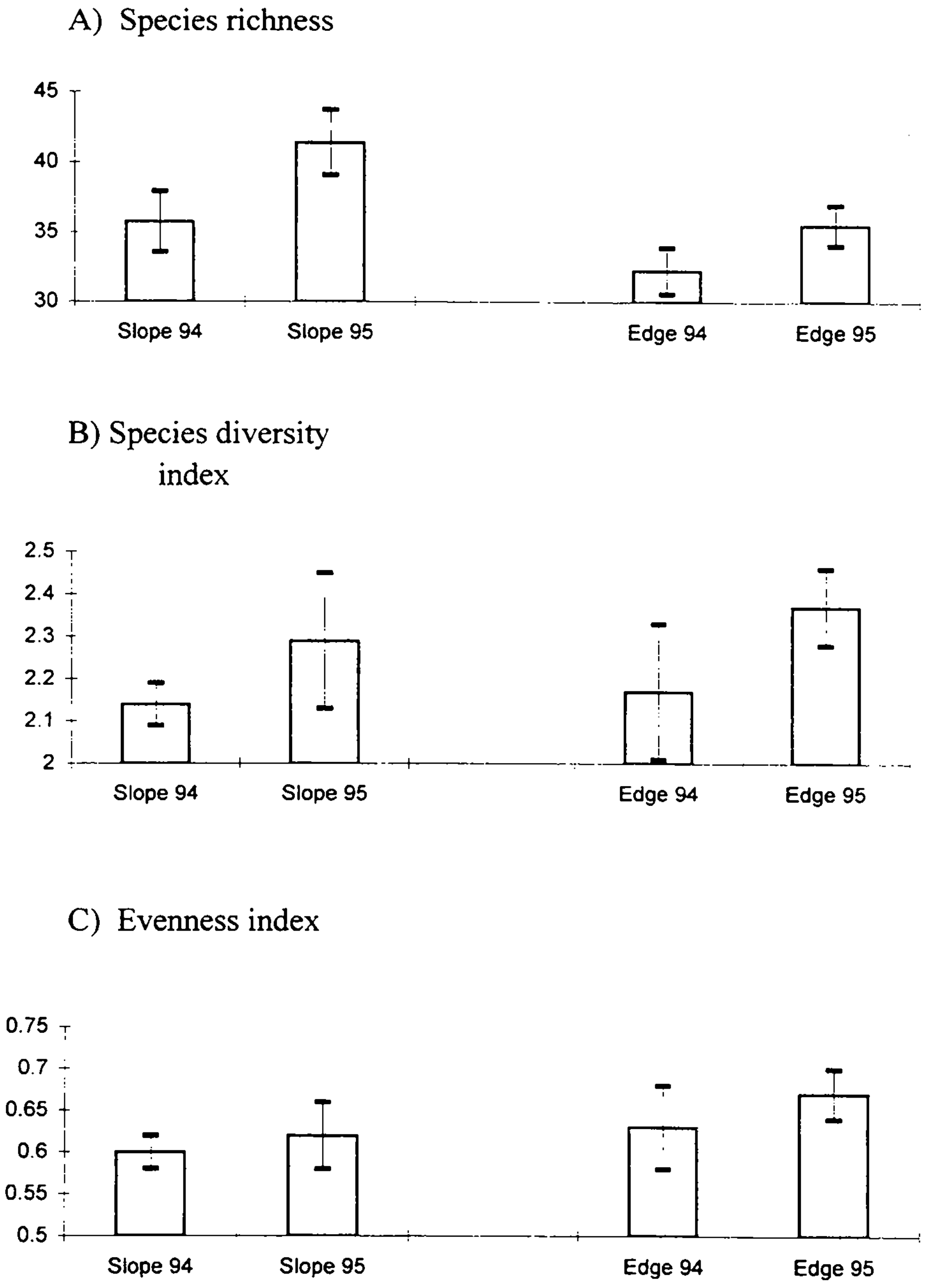


Fig. 4.4. Temporal pattern of total species richness of three families of coral reef fishes: Chaetodontidae, Labridae, and Pomacentridae from two habitats (A-reef slope, B-reef edge) at eight localities of Phuket surveyed in 1994 and 1995.


A) Species richness

B) Species diversity index

C) Evenness index

Fig. 4.5. Temporal variation of fish community measures (mean ± SE): A) Species richness, B) Species diversity index, and C) Evenness index, from two habitats at eight localities of Phuket surveyed in 1994 and 1995.

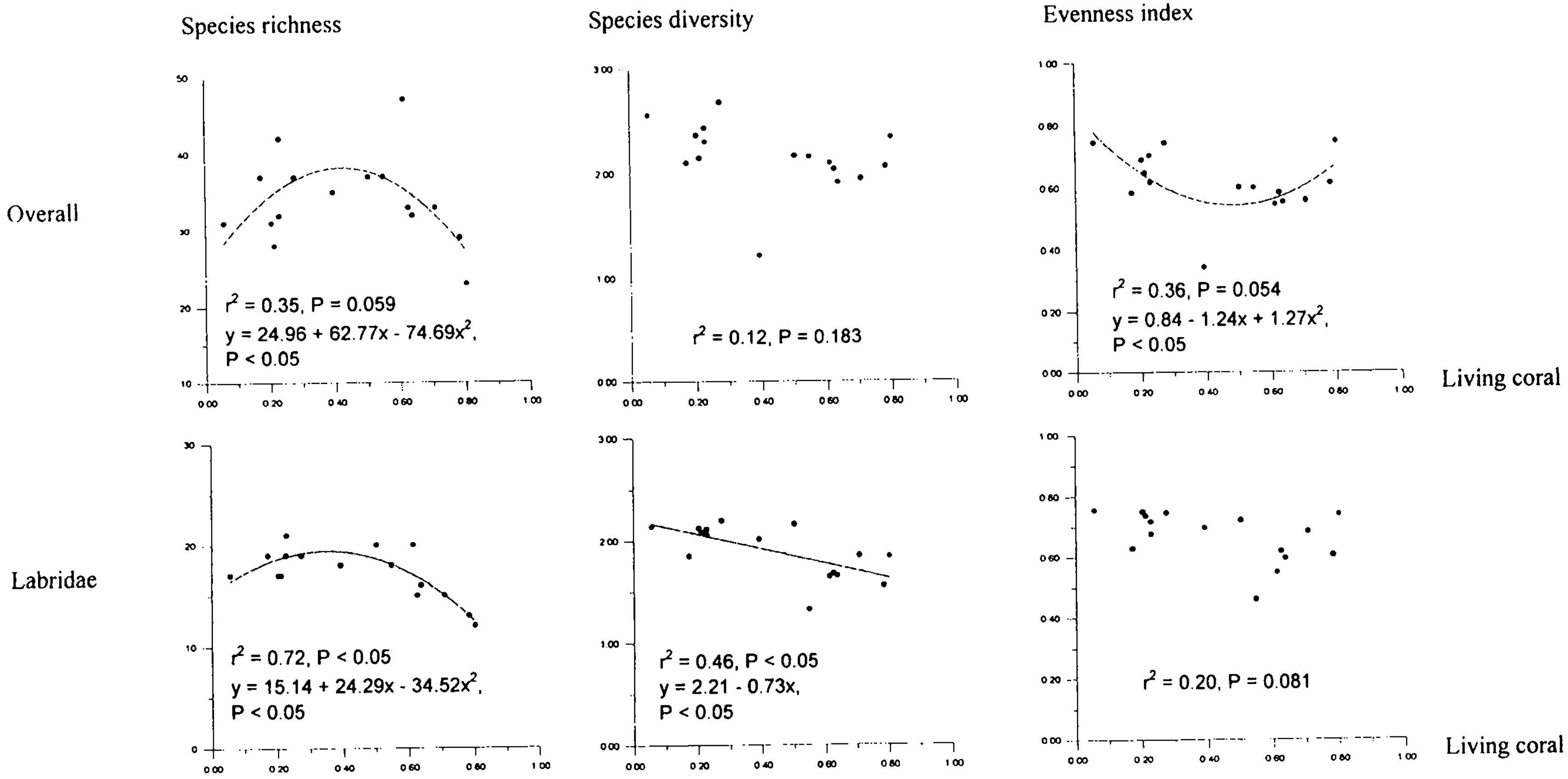


Fig. 4.6. Relationships between living coral cover (proportion) and community parameters of some coral reef fish assemblages at Phuket, Thailand during March 1994.

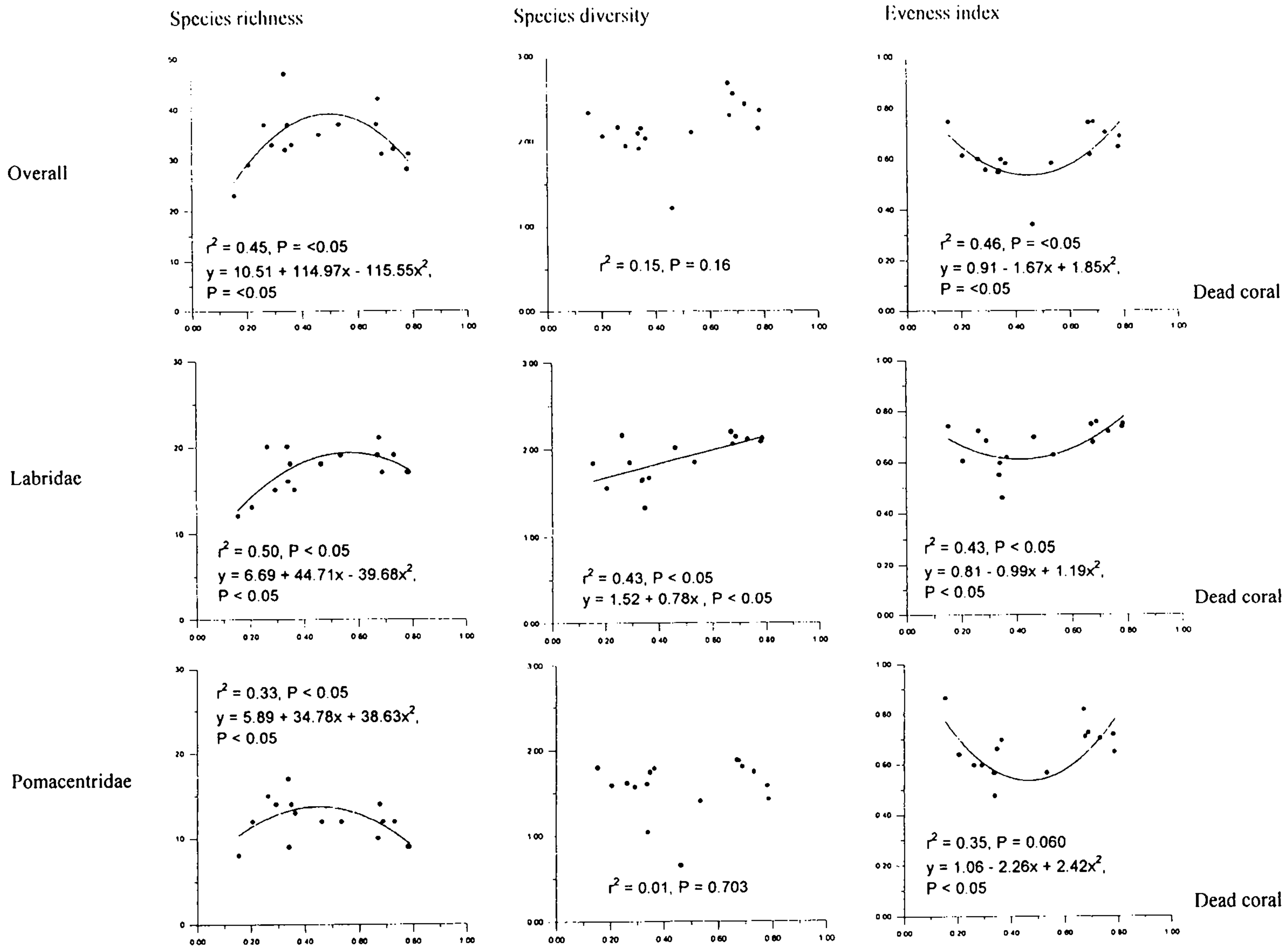


Fig. 4.7. Relationships between dead coral cover (proportion) and community parameters of some coral reef fish assemblages at Phuket, Thailand, during March 1994.

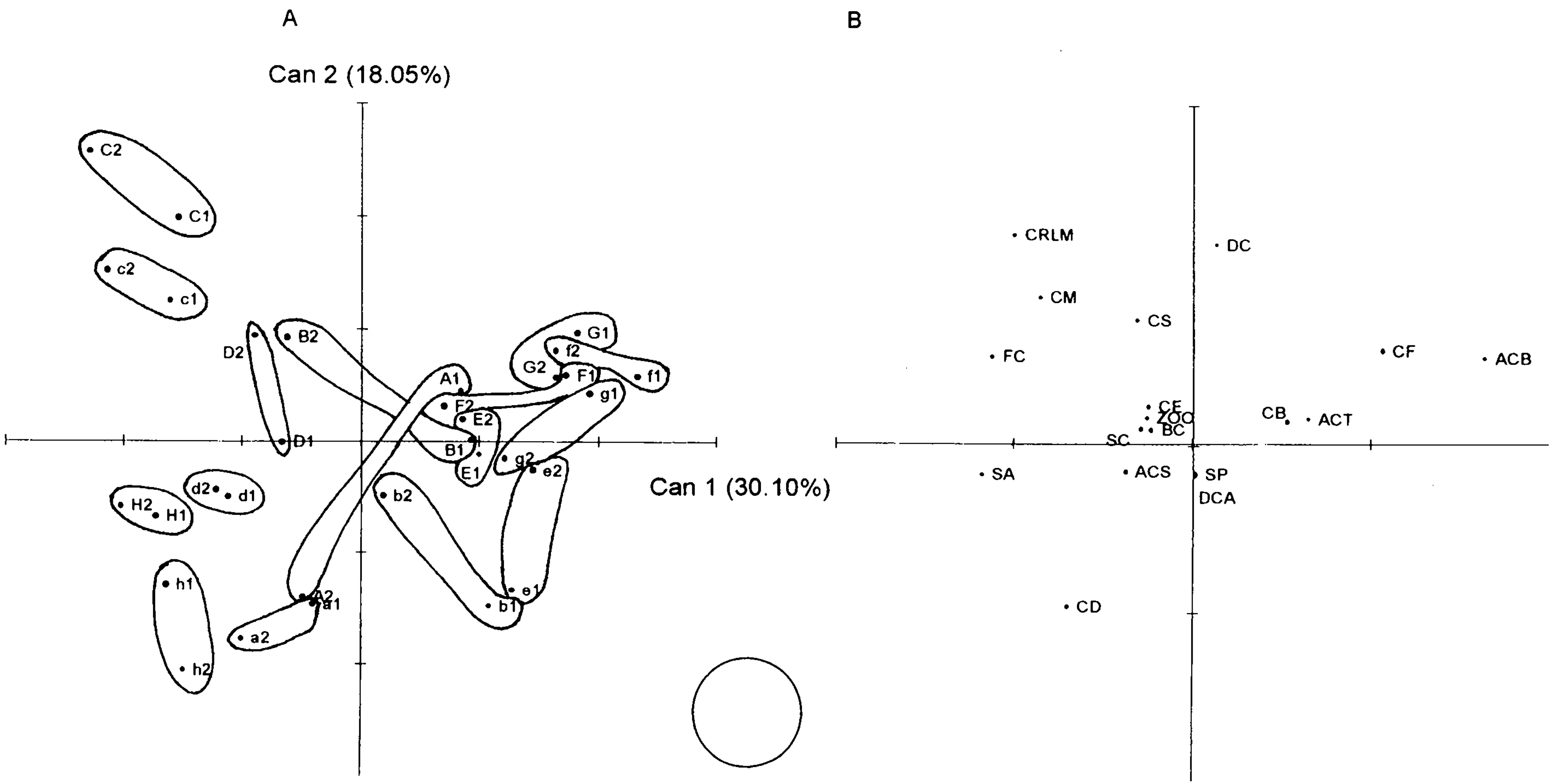


Fig. 4.8. Ordination plots of CDA: A) Sites and B) Canonical structure, showing the spatio-temporal variation of the benthic assemblage within two habitats at eight localities at Phuket, Thailand, surveyed in 1994 and 1995. Site code name is represented by letters as described in Fig. 4.1 (capital letter = reef slope, small letter = reef edge) and numbers represent two surveys (1 = March 1994 and 2 = March 1995). Benthic lifeforms are defined in Appendix 4.1. Circles indicate 95% confidence limit around group centroid.

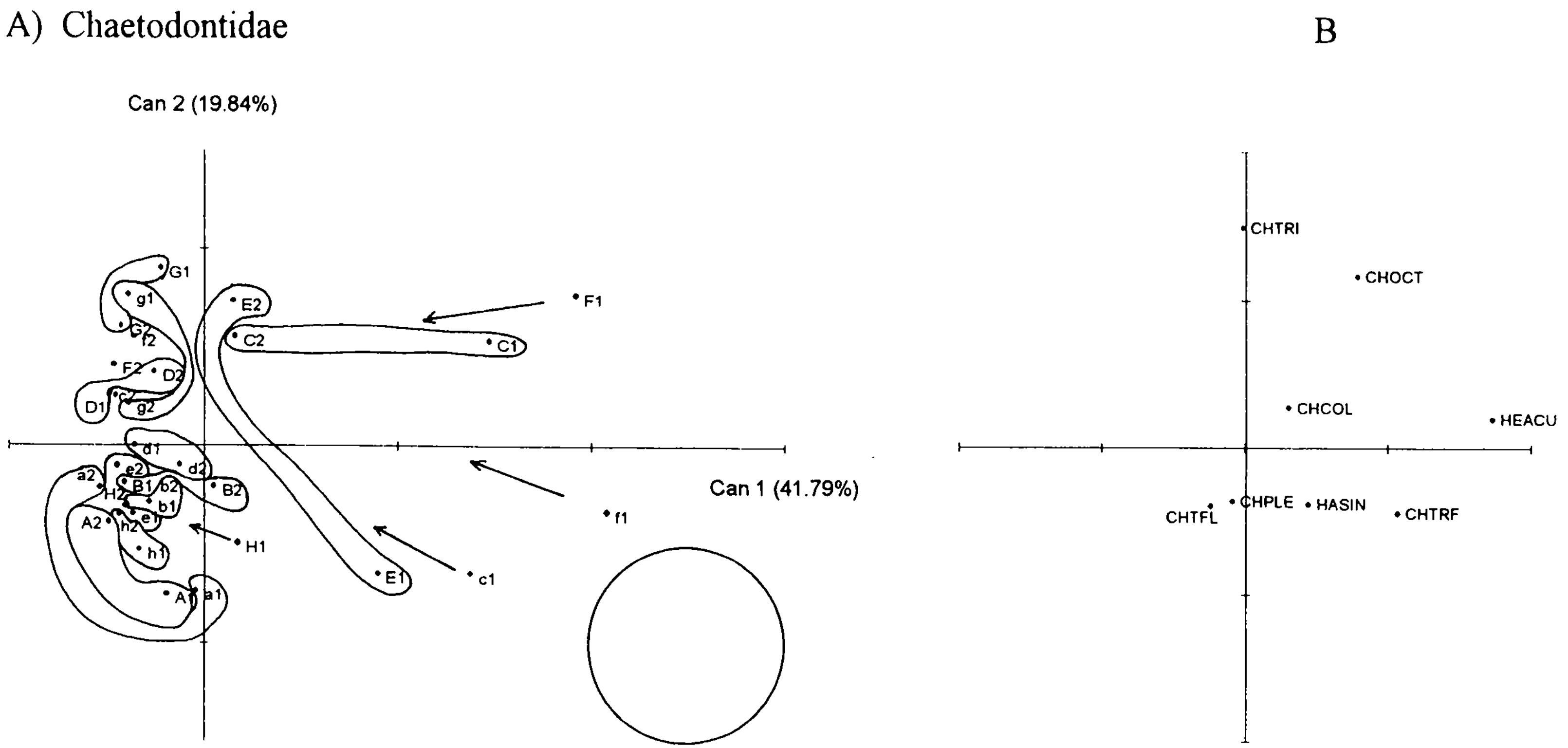


Fig. 4.9A. Ordination plots of CDA: Sites canonical scores (left) and Fish canonical structure (right), showing spatio-temporal variation of the assemblage structure of chaetodontid fishes from two habitats at eight localities each at Phuket, Thailand, surveyed in 1994 and 1995. Site code name is as described in Fig. 4.8 and fish codes in Appendix 4.2. Circle indicates 95% confidence limit around each group centroid.

## B) Labridae

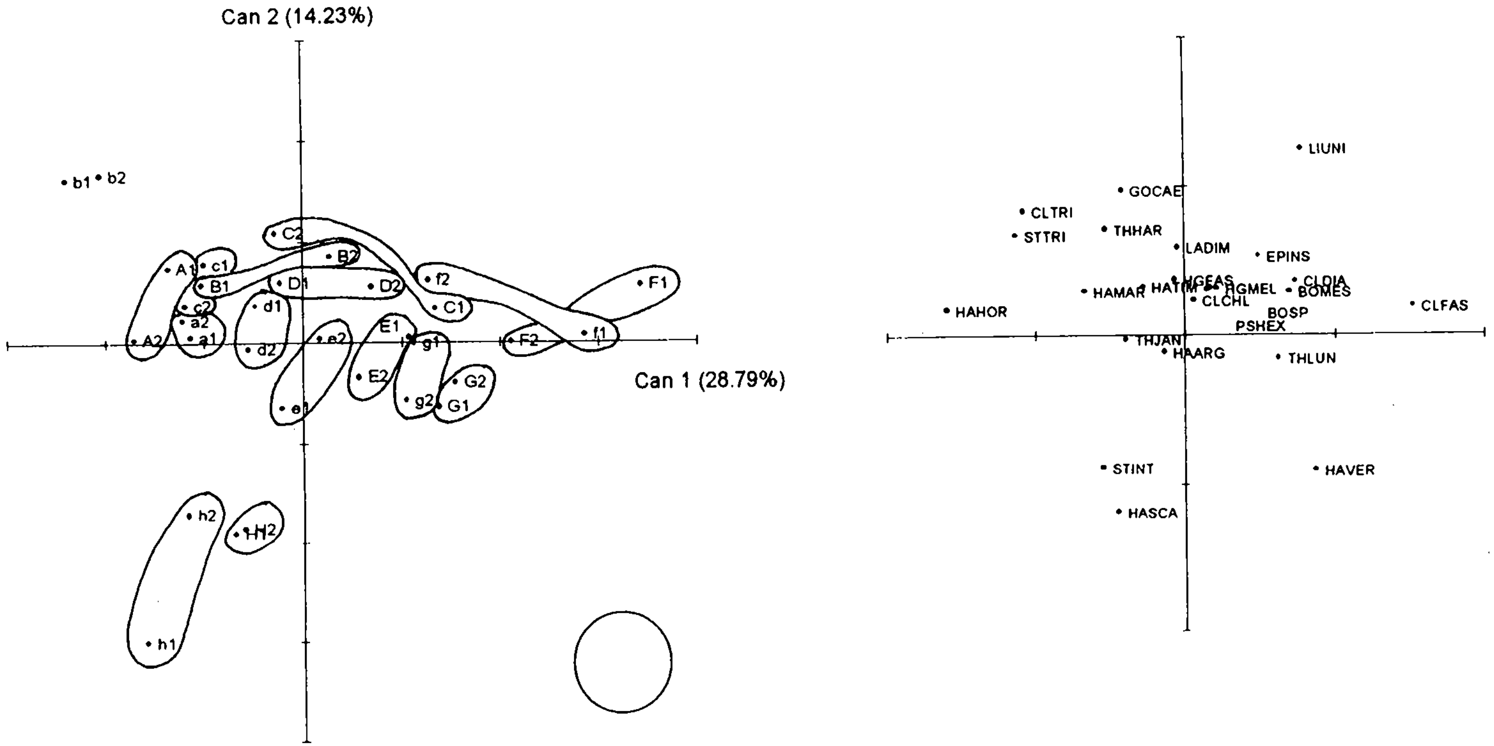


Fig. 4.9B. Ordination plots of CDA: Sites canonical scores (left) and Fish canonical structure (right), showing spatio-temporal variation of the assemblage structure of labrid fishes from two habitats at eight localities each at Phuket, Thailand, surveyed in 1994 and 1995. Site code name is as described in Fig. 4.8 and fish codes in Appendix 4.2. Circle indicates 95% confidence limit around each group centroid.

## C) Pomacentridae

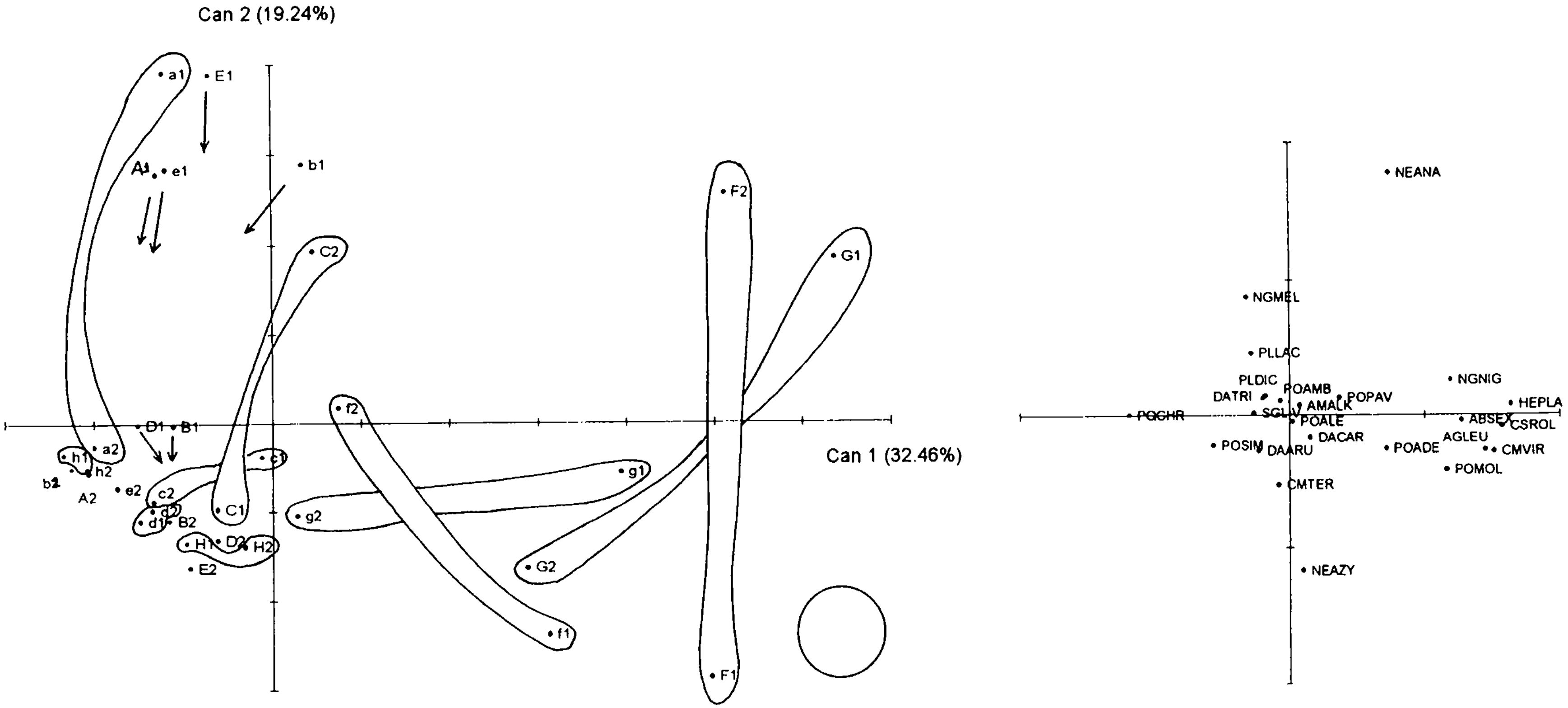


Fig. 4.9C. Ordination plots of CDA: Sites canonical scores (left) and Fish canonical structure (right), showing spatio-temporal variation of the assemblage structure of pomacentrid fishes from two habitats at eight localities each at Phuket, Thailand, surveyed in 1994 and 1995. Site code name is as described in Fig. 4.8 and fish codes in Appendix 4.2. Circle indicates 95% confidence limit around each group centroid.

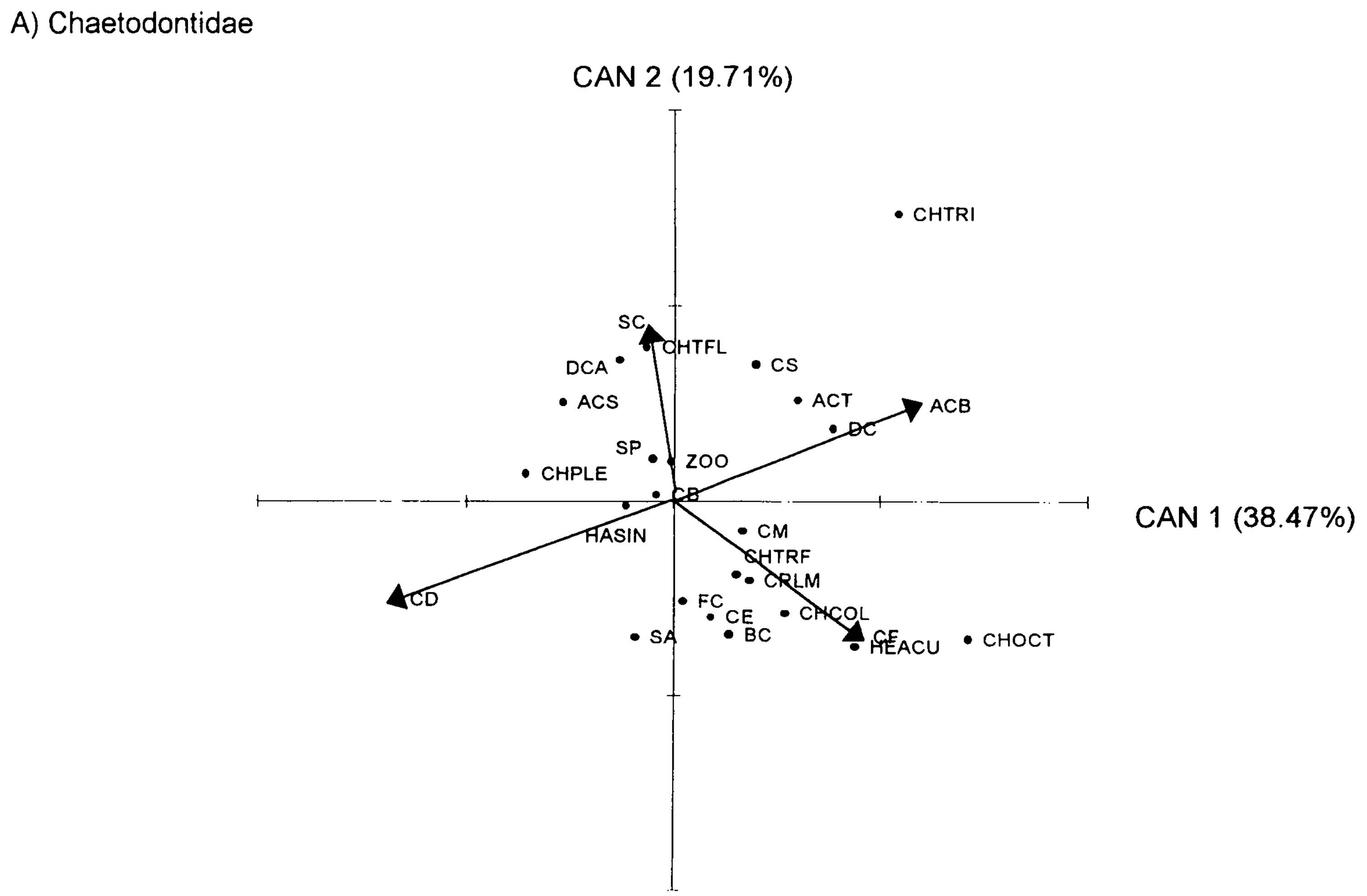


Fig. 4.10A. Results from CCA showing trends in the relationship between habitat structure and chaetodontid fishes. Arrows indicate the direction of trends based on benthic lifeforms but only potential trends are presented (canonical structure usually higher than 0.5). Benthic lifeforms and fish species are defined in Appendix 4.1 and 4.2.

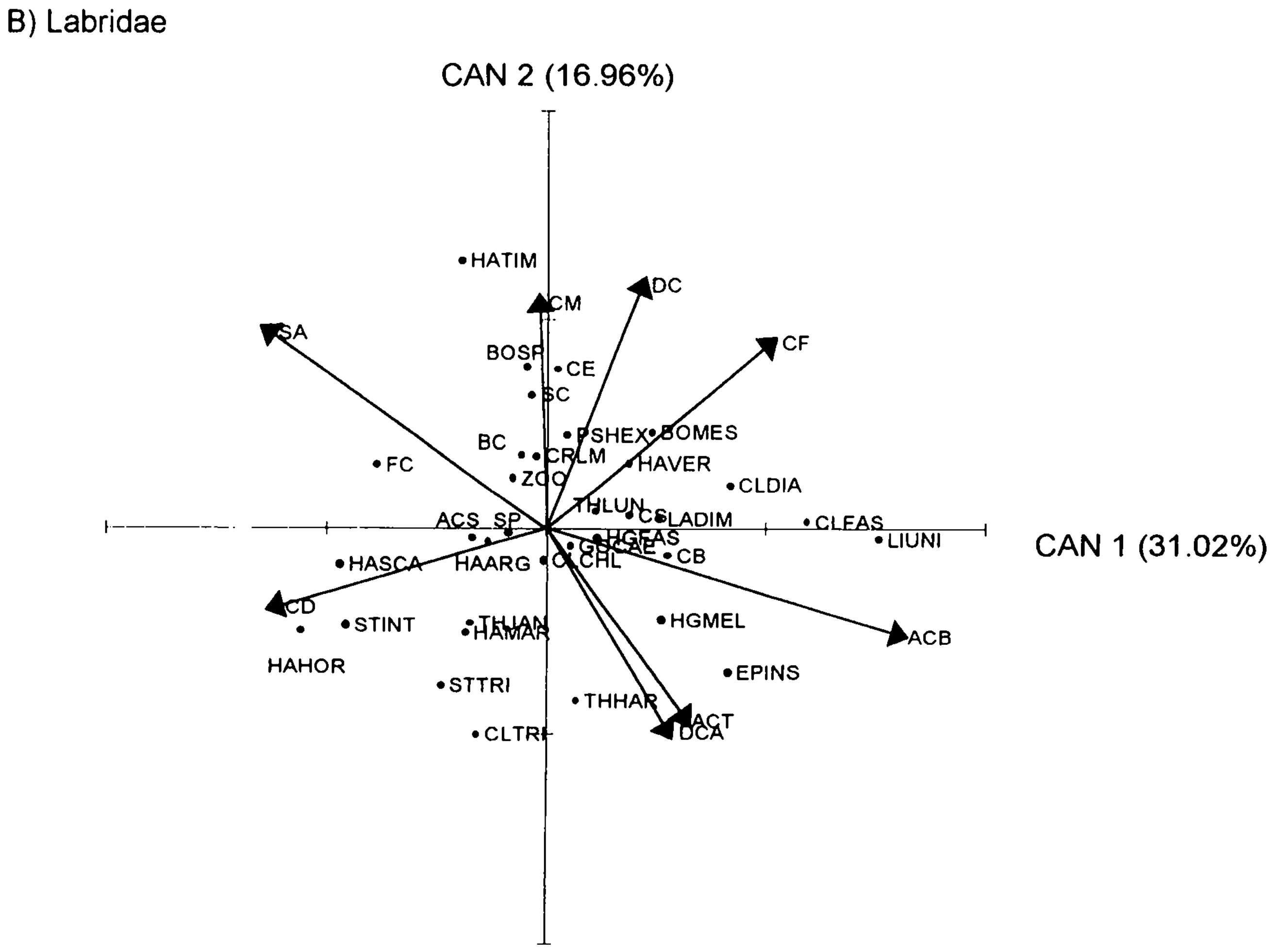
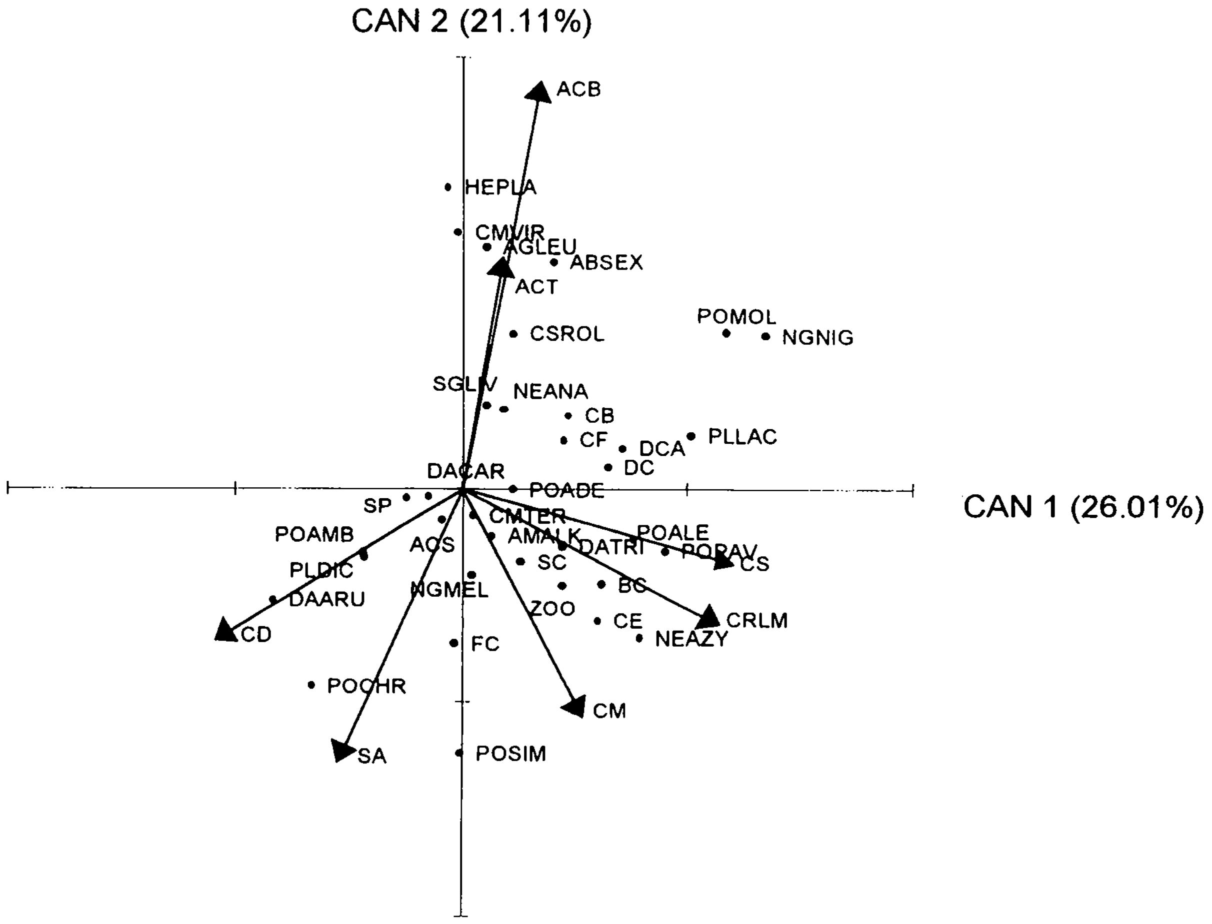


Fig. 4.10B. Results from CCA showing trends in the relationship between habitat structure and labrid fishes. Arrows indicate the direction of trends based on benthic lifeforms but only potential trends are presented (canonical structure usually higher than 0.5). Benthic lifeforms and fish species are defined in Appendix 4.1 and 4.2.



C) Pomacentridae

Fig. 4.10C. Results from CCA showing trends in the relationship between habitat structure and pomacentrid fishes. Arrows indicate the direction of trends based on benthic lifeforms but only potential trends are presented (canonical structure usually higher than 0.5). Benthic lifeforms and fish species are defined in Appendix 4.1 and 4.2.

# CHAPTER 5

# RELATIONSHIPS BETWEEN LABRID FISHES AND HABITAT STRUCTURE ON TWO NEAR-SHORE REEFS OF THE CENTRAL GREAT BARRIER REEF

## 5.1 SUMMARY

The relationships between labrid fish assemblages and habitat structure (coral reef benthic assemblages) were investigated on the fringing reefs of two near-shore islands of the central Great Barrier Reef (Orpheus and Magnetic). Two habitats (middle reef slope and reef edge) were surveyed at each of 13 study sites. The quantitative relationships between wrasse and benthic assemblages were examined by considering both univariate community parameters (species richness, evenness, diversity and overall abundance or % cover) and multivariate approaches. Measures of diversity and total abundance were not particularly sensitive indicators of differences between the two islands or two habitat types, but indicated that there was considerable variation among locations. Fish species richness, diversity, evenness and total abundance tended to be negatively related to living hard coral and algal cover, but exhibited positive relationships with dead coral and soft coral. However, fitted linear or quadratic relationships explained little variation (10-30%). More distinctive patterns were revealed by consider community structure. Both habitat structure and wrasse assemblages showed a similar trend in that there was a greater influence of large scale spatial variation over broad and fine scales (island > locality > habitat). Multivariate analysis revealed significant correlations between each particular benthic lifeform and each species of wrasse (e.g. *Halichoeres dussumeri* and *Halichoeres biocellatus* were found where reefs were covered with macroalgae, foliose coral, encrusting coral, and tabulate *Acropora; Thalassoma lunare* and *Labroides dimidiatus* were found at sites dominated by soft coral; and *Halichoeres melanurus* was found associated with dead coral). The nature of this relationship, in general, tends to be predictive (meaning that the presence/abundance of a particular benthic lifeform can indicate abundance of labrid species) for habitat structure and responsive (meaning that the

presence/abundance of a particular species of labrid is predictable by habitat structure) for labrids.

The temporal patterns in wrasse assemblage were also studied during a two year period at three localities on Orpheus Island. Community parameters indicated some seasonal variation but with overall annual stability, and spatial patterns persisted over time. Results confirm that wrasses represent good indicators of change in a wide range of coral, algal and abiotic substrata on coral reefs.

## 5.2 INTRODUCTION

Coral reefs provide a wide range of habitats for a great diversity of fish assemblages. Fish-habitat relationships have been extensively studied in many regions of the world in an attempt to explain the processes controlling the abundance and distribution of fish (Bell & Galzin, 1984; Roberts & Ormond, 1987; Roberts *et al.*, 1992). In addition, linkages between fish and coral habitat communities have been sought, with the goal of using fish as indicators of reef status (Reese, 1972; Luckhurst & Luckhurst, 1978; Findley & Findley, 1985) or the effects of disturbance (McClanahan, 1994). Many of these studies have focussed on butterflyfishes (family Chaetodontidae) and their response to changes in coral cover alone, with contrasting results (Reese, 1981; Bouchon-Navaro, *et al.*, 1985; Roberts & Ormond, 1987; Roberts, *et al.*, 1988). Despite their diversity, butterflyfishes exhibit a relatively narrow range of feeding modes and are not conspicuously abundant in most reef habitats. Recent studies suggest that wrasses (family Labridae), may be more sensitive to a wider range of changes to habitats, because of their greater diversity, abundance and range of habitat requirements (Jones & Kaly 1995, Chapter 4). An investigation of quantitative relationships between wrasses and habitats across a range of spatial and temporal scales, will provide more understanding of the factors and processes which control the abundance and distribution of this potentially useful group of indicators.

On the Great Barrier Reef, there have been few studies to assess the quantitative relationships between habitat and fish assemblages (McCormick, 1994, 1995; Green, 1996). Most efforts have been made towards describing the patterns of distribution and abundance of either fish or benthic assemblages, with little attempt to integrate the findings. These studies have been conducted at various spatial scales from large - between reef (Done, 1982; Williams, 1982; Williams & Hatcher, 1983; Dinesen, 1983; Russ, 1984a, 1984b; Bradbury *et al.*, 1986), to broad and fine scales - within reef and within habitat (Fowler, 1990; Meekan *et al.*, 1995). Discussion of the processes influencing distribution and abundance over different spatial scales, have usually focused on the recruitment of the reef organisms (Done, 1982, 1983; Williams, 1982, 1991). However, deterministic relationships between coral reef fish and their habitat may indicate other processes and underlying mechanisms that structure fish assemblages on the Great Barrier Reef. While a few studies have investigated the relationship between fish and habitat variables (McCormick, 1994, 1995; Green, 1996), they have primarily focused on relatively pristine off-shore reefs. There has been little attention paid to near-shore reefs, which are more likely to be subject to habitat degradation.

The near-shore reefs of the Great Barrier Reef are important in terms of ecological and socio-economical values. They are situated close to the land and as exploitation, pollution and nutrient enrichment from the land exists, they are likely to be impacted first and most. Therefore, these reefs may represent early warning stations for impacts on the reef as a whole. To date, coastal enrichment and increased sedimentation is still a controversial issue for the Great Barrier Reef (Bell, 1991; Walker, 1991; Kinsey, 1991; Steven & Larkum, 1993). As pointed out by Kinsey (1991) there is no conclusive evidence that it is or is not a problem threatening coral reef systems. The information on water quality alone (e.g. nutrient load) is not sufficient for indicating the impact of land pollution on coral reefs. The application of bioindicators might be an alternative way to detect any impact and is being increasingly adopted as an important tool for environmental quality assessment (Jones & Kaly, 1996). While the application of a single indicator is usually limited to illustrate the impact from disturbance (Harding, 1992), the application of multiple indicators or a group of indicators (same guild or same taxonomic group) is an alternative. Establishing which fish taxa is a good indicator for habitat changes requires a detailed analysis of fish-habitat relationships.

The objectives of this study were to investigate the relationships between habitat structure and labrid fish assemblages, with an emphasis on detecting features of the habitat that explain patterns of distribution, abundance and community composition.

The study investigates the relationships between benthic lifeforms and wrasse assemblages on near-shore coral reefs in the central Great Barrier Reef. The nature of the relationships was examined across multiple spatial scales: (1) Variation among reefs associated with two inshore islands, ca. 100 km apart; (2) Variation among locations within islands; and (3) Variation among habitats within each location. Locations around a single island were monitored in order to assess the seasonal and annual variation in community structure that was independent of changes to habitats.

## 5.3 MATERIALS AND METHODS

### 5.3.1 STUDY AREA AND DESIGN

The study was carried out at two near-shore islands on the central section of the Great Barrier Reef (18° S to 19° 10′ S, 146° E to 147° E), both within 20 km from the coast (Fig. 5.1): Orpheus Island (15 km) and Magnetic Island (5 km). Description of spatial patterns was based on a total of 13 localities monitored between November 1993 and January 1994. The number of study localities on each island varied according to the degree of reef development, and included nine localities at Orpheus Island and four at Magnetic Island. Two habitats, the middle reef slope and reef edge, were examined at each locality. The reef slope was examined at a depth of 6 - 8 m below mean sea level (MSL) whereas the reef edge was 3 - 4 m below MSL. The term "study site" is used to refer to each habitat at each locality.

An examination of temporal pattern in labrid assemblages was carried out on the reef slope and reef edge at three localities on Orpheus Island: Cattle Bay, North Pioneer Bay and South Pioneer Bay. The data were collected every 2 - 3 months (10 times in total) from December 1993 to December 1995.

### 5.3.2 SAMPLING TECHNIQUE

The benthic coral-reef assemblage was studied using a Line Intercept Transect technique (modified from Loya, 1978) in which benthic lifeforms (modified from Dartnall & Jones, 1986) were used for classification instead of taxonomic groups. This was because the aim of the study concentrated on the effects of habitat structure on fishes rather than the benthic assemblage structure itself. The labrid assemblage was quantified using the instantaneous visual census technique (Fowler, 1987). Sampling of both fish and benthic lifeforms was based on the same 30 m line transects with six replicates per site. For the fish censuses, wrasses within 2.5 m from each side of the transect (transect dimension equals 5x30 $m^2$) were counted, identifying each individual to species level. Wrasses recorded were further categorised into three reproductive stages: terminal, initial and juvenile phases. The sampling scheme was based on the optimization procedure carried out at Orpheus Island (Chapter 3). Transect lines were laid out parallel to the reef and as far as possible at the same depth. Replicate transects were laid at random in the same habitat, but with each transect separated by at least 30 m.

### 5.3.3 ANALYSIS

The variables used for calculate community parameters of both habitat structure and wrasse assemblage were: mean area cover (%) of each benthic lifeform, mean abundance of wrasses (individuals per 150 $m^2$), number of lifeforms/fish species (richness) recorded at each site, occurrence of each lifeforms/fish species across all sites, total area cover/fish abundance of each lifeform/fish species across all sites. Species diversity index (H′) evenness index (E), and habitat width index (AH′) was calculated. Mean area cover/fish abundance was used to calculated a Shannon-Wiener index of diversity ($H' = -\Sigma p_i \log_e p_i$) and evenness index ($E = H'/H_{max}$) (Pielou, 1974; Magurran, 1988) both for site (species diversity) and for species (site diversity). Site diversity was thus used to calculate habitat width ($AH' = 2.7183\ H'$, Pielou, 1974; Fowler, 1990). The relationship between habitat structure and wrasse assemblage was considered: 1) between benthic and fish community parameters and 2) between area cover of major lifeforms and community parameters of wrasse. A series of linear and quadratic regression analyses (Zar, 1984) was carried out to describe the nature of bivariate relationships between fish community parameters and habitat variables.

*A priori* tests on both habitat structure and wrasse raw data were performed to satisfy the assumptions of parametric statistical methods, both for univariate (Underwood, 1981) and multivariate (Bray & Maxwell, 1982) analyses. In most cases, double square-root transformation was appropriate for habitat structure data while log (x+1) transformation was appropriate for the wrasse data. Both transformations were also used to reduce the chance that a few extremely dominant lifeforms/fish species would dominate in the multivariate statistical analysis. For all statistical hypothesis tests, a significance level of 0.05 was used, but a lowest level was reported when appropriate.

For multivariate analyses, data reduction was necessary as there was some rare variables present in the data matrix in which reduces the power of multivariate statistical analysis (Tabachnick & Fidell, 1989). Data reduction was carried out by omitting very rare variables or those that occurred at less than 20% of all sites (< 5 sites).

A Multivariate Analysis of Variance (MANOVA) was used to test for significant differences in community composition among locations, sites and/or times. The Pillai's trace *criterion* was used in a MANOVA because of its robustness over a wider range of conditions (Green, 1979). MANOVA model can be generated in the same way as an ANOVA model (Brays & Maxwell, 1982). The model used for the analysis of spatial pattern was based on two fixed factors: locality (13 levels) and habitat (2 levels). An unbalanced mixed model with three factors: Island group (2 levels), locality (13 levels unbalanced nested within group) and habitat (2 levels fixed factor) was not used because the zero degreess of freedom caused by Island group factor. However, the effect of Island group can also be examined using canonical discriminant analysis.

For the temporal study, the analysis model used was based on three fixed factors: time (10 levels), locality (3 levels) and habitat (2 levels). Because the data at some localities was not collected ten times this caused an unbalanced design. Therefore, the MANOVA was based on a type III Sum of Squares and Cross Products (SS&CP) matrix. The analysis was executed by Proc GLM in SAS (SAS Institute Inc., 1990).

Canonical Discriminant Analysis (CDA) was employed as an *a posteriori* test following the MANOVA to help identify the nature of any significant differences detected. CDA was performed on a transformed and centred data matrix, which was executed using Proc CANDISC in SAS (SAS Institute Inc., 1990). The canonical structure of each variable was used as a responsive factor for the discriminant. Angular interpretation was thus used to interpret the ordination plot produced by CDA.

Canonical Correlation Analysis (CCA) was performed to determine the relationship between habitat structure and wrasse assemblage. Double square root transformed data for habitat and log (x+1) transformed data for wrasse were used as they were centred prior to analysis. The canonical intraset structure (correlations) was used to illustrate general trends of relationships between fish and habitat structure variables. Redundancy analysis was also carried out to show how much of the variation in the fish variables/data set could be explained by the habitat structure variables/data set as well as by their own variables/data set. In the same way, the habitat structure variables/data set could be explained in terms of the fish variables/data set and their own. The analysis was performed using Proc CANCORR in SAS (SAS institute Inc., 1990).

## 5.4 RESULTS

### 5.4.1 DOMINANT COMPONENTS OF THE HABITAT AND LABRID FISH FAUNA

A total of 20 lifeforms (Appendix 5.1) and 36 species of wrasse (Appendix 5.2) were recorded from reef edge and reef slope of Orpheus and Magnetic Islands. In terms of relative occurrence, 16 different lifeforms and 17 different wrasses were found more than 20% of all sites (Appendix 5.1 and 5.2).

Habitat structure on the inshore reefs of both islands were dominated by four lifeforms; soft coral, dead coral, macro algae, and coral debris. Each accounted for at least 10% of the cover, and the combination of all four categories accounted for approximately 55% of area cover by all benthic categories (Appendix 5.1). The composition of benthic assemblage between Magnetic and Orpheus Islands in general, was clearly different (Fig. 5.2A). At Magnetic Island, the benthic assemblage was dominated by macroalgae, foliose coral and encrusting coral. In contrast, dead coral, soft coral, coral debris and massive coral dominated at Orpheus Island.

The overall wrasse assemblage was dominated by three species: *Halichoeres melanurus, H. dussumeri,* and *Thalassoma lunare,* which represent *ca* 55% of all fish recorded (Appendix 5.2). The composition of the wrasse assemblage between Magnetic and Orpheus Islands can also be distinguished (Fig. 5.2B). At Magnetic Island, the wrasse assemblage was dominated by *Halichoeres dussumeri* and *Halichoeres biocellatus* while at Orpheus Island, *Halichoeres melanurus, Thalassoma* *lunare, Hemigymnus melapterus* and *Labroides dimidiatus* dominated the whole assemblage.

### 5.4.2 SPATIAL PATTERN

The community parameters of habitat structure: lifeform richness, lifeform diversity and evenness, were not significantly different between reef slope and reef edge (Z = 1.25, 0.94, 0.94 and P = 0.213, 0.345, 0.345 respectively, Wilcoxson paired-sample test). The community parameters for wrasse assemblage, including mean total abundance were also not significantly different among the two habitat zones (Z = 0.36, 0.25, 0.52, 1.22 and P = 0.721, 0.807, 0.600, 0.221 respectively). Therefore the average of these parameters between habitats was used to describe differences among islands and localities (Fig. 5.3). The community parameters of habitat structure indicated little difference between Orpheus and Magnetic Islands, but some variation among locations (Fig. 5.3A). Mean lifeform richness ranged between 10-14 lifeforms, mean lifeform diversity ranged between 1.40 - 2.00, and the mean evenness index ranged between 0.65 - 0.75.

There was some variation for the results of community parameters of wrasse assemblage (Fig. 5.3B). The Orpheus Island localities (excepted O4 and O7) tended to exhibit higher values of species richness, diversity, evenness and total abundance than those at Magnetic Island. Species richness ranged between 13 - 15 species at Orpheus Island 7 - 10 species at Magnetic Island. The species diversity index ranged between 1.60 and 2.10 at Orpheus and was in the range of 0.80 -1.60 at Magnetic Island. Similarly, the evenness index was 0.70 - 0.85 at Orpheus Island and 0.40 - 0.70 and Magnetic Island.

The MANOVA of both habitat structure and the wrasse assemblage detected significant interactions between locality and habitats (Table 5.1). This suggested inconsistent differences between reef slopes and reef edges across all localities. Canonical discriminant analyses based on site classification were therefore used to illustrate the nature of these significant differences. The CDA's results of the first three canonical discriminants of habitat structure explained 77.5% of total variance and the first four of the wrasse assemblage explained 82.6%. Each of the remaining canonical discriminants explained less than 5% of the total variance.

The ordination plots of the first two canonical discriminants of both habitat structure (explaining 71.8% of the variation) and the wrasse assemblage (70.4%) indicated a concordant pattern between fish and habitat. Axis one (explaining most of the variation) separated the sites into two distinct groups, representing distinct community structures at Magnetic and Orpheus islands (Fig. 5.4). Ordination plots showed that the habitat structure of Magnetic Island was characterised by macroalgae, foliose coral, encrusting coral and tabulate *Acropora*, while Orpheus Island was characterised by soft coral, dead coral, massive coral, fire coral, coral debris and branching coral (Fig. 5.4A). The wrasse assemblage at Magnetic Island was characterised by *Halichoeres dussumeri, H. biocellatus, Choerodon graphicus* and *Cheilinus trifasciatus* while Orpheus Island was represented by *Labroides dimidiatus, Thalassoma lunare, Halichoeres melanurus* and *Cheilinus fasciatus* (Fig. 5.4B).

There was some differences among sites within the two islands in the patterns of fish assemblage structure and habitat structure (Fig. 5.4). Habitat structure tended to be very uniform at Magnetic Island, with both locations and depth strata being very similar. The reef slope and reef edge at Orpheus Island was different at most localities, except at O2 and O5 (Fig. 5.4A). Axis two identified a mixture of differences among locations and depth strata, with no obvious groups or trends. The reef slope and reef edge of O1 and O3 were characterised by greater cover of soft coral, dead coral and branching *Acropora*, which were more common on the reef edge than on the reef slope, and dead coral with algae which dominated the reef slope. At locations O4, O6, and O7, the reef slopes had more soft coral than the reef edge. The reef edge at O8 and O9 had more soft coral and branching *Acropora*, and less coral debris and dead coral than on the reef slope.

For wrasses, there was no consistent difference the between the reef slope and reef edge at Orpheus Island, while at Magnetic Island, these zones could be distinguished (Fig. 5.4B). The reef slope of M1, M2, M3, and M4 had more *Halichoeres melanurus* and *H. nebulosus*, but less *Hemigymnus melapterus* than on the reef edge.

### 5.4.3 WRASSE-HABITAT RELATIONSHIPS

The bivariate relationships between the community parameters used to describe habitat structure and the wrasse assemblage varied, depending on the variables used.

The area cover of four major lifeforms; living coral, dead coral, algae, and other fauna, showed significant relationships with most community parameters of wrasse, although the variance explained was not so high (Fig. 5.5). The nature of most relationships if detected, appeared to be linear (Fig. 5.5). The species richness, diversity, evenness and total abundance of wrasses tended to be inversely related to living coral cover, although significant parabolic relationships for evenness and total abundance suggested that the relationship was positive at some levels of coral cover. However, there was generally more species and a more similar relative abundance of species at low coral cover. There was also a tendency for these parameters to be negatively related to algal cover, although this was only statistically significant for diversity and evenness. In contrast, there was a tendency for positive linear relationships between fish species richness, diversity, evenness and total abundance, both for dead coral and other fauna (including soft coral).

The results of the CCA showed a significant correlation between multivariate descriptors of habitat structure and wrasse assemblage, with the first three canonical variates explaining 85.1% of total variance (Table 5.2). Redundancy analysis indicated that the amount of variation in the canonical axes explained by both fish and habitat was moderate, the first three canonical variates accounted between 36.2-53.1% of the variation. There was asymmetry in the degree to which fish and habitat contributed to the canonical correlation. Habitat structure had predictive relationships to the wrasse assemblage (with habitat structure explaining 42.6% of the variation of the wrasse assemblage), while the wrasse assemblage was primarily "responsive" to habitat structure (ability of wrasse assemblage to explain habitat structure is 36.2%). Intraset relationships (coexistence) were slightly more predictive for the wrasse assemblage (53.1%) than habitat structure (47.1%), indicating that the co-existence among wrasse species in the assemblage is slightly higher than the co-existence of benthic lifeforms.

The results of the redundancy analysis also provided information on the contribution of each variable from both data sets to the combined axes, that is, highly correlated features of both fish and habitat assemblages. For habitat structure, the important variables were: macro algae, foliose coral, soft coral, encrusting coral, dead coral, massive coral, tabulate *Acropora*, coral debris, sand and fire coral. For the wrasse assemblage the species were: *Halichoeres dussumeri, Labroides dimidiatus, Thalassoma lunare, Halichoeres melanurus* and *Halichoeres biocellatus.*

The ordination plot of the first two canonical variates (79.8%) showed the main trends in the relationships between variables of two data sets (Fig. 5.6). When interpreted in conjunction with the redundancy analysis, the relationships between a specific fish species and habitat variable can be seen. *Halichoeres dussumeri* and *Halichoeres biocellatus* appear to be found where reef are covered with macro algae, foliose coral, encrusting coral, and tabulate *Acropora*. *Thalassoma lunare* and *Labroides dimidiatus* appeared to be found at sites dominated by soft coral, and *Halichoeres melanurus* was mainly found associated with dead coral.

### 5.4.4 TEMPORAL VARIATION OF WRASSE ASSEMBLAGE

Two wrasse species, *Halichoeres melanurus* and *Thalassoma lunare* dominated the Orpheus Island study sites monitored over a two-year period. These two species represented 64% of the combined abundance of all species, *H. melanurus* alone accounting for 51%. During study of the temporal variation , a total of 28 species were recorded from three localities on Orpheus Island over the two years period (Appendix 5.3). In terms of relative occurrence, 18 species of wrasses were found at more than 20% of all sites (Appendix 5.3). Statistical analyses were restricted to these species.

There were no significant differences between reef slope and reef edge habitats, in terms of species richness, species diversity and total abundance over the 10 sampling times (Appendix 5.3) ($Z =$ 1.26, 0.73, 2.55 and $P =$ 0.207, 0.466, 0.072 respectively, Wilcoxson paired-sample test). The evenness index exhibited a marginal significant difference ($Z = 1.98$ and $P = 0.047$). However, the average between two habitats of these parameters were used to describe general patterns across time.

There were some weak seasonal patterns in total abundance, species richness, evenness and diversity at the three localities (Fig. 5.7). Commonly, the values of all parameters were likely to decrease during April and lowest around July. These then increased rapidly and maintained high values over the summer months. However, the relative range of variation differed among parameters. The evenness index was less variable indicating that the relative abundance of most species was maintained over time. This result explained the very similar patterns for species richness and species diversity, which appear to be driven by the seasonal inclusion and loss of species. Seasonal variation appeared to be more pronounced that different among the two years, perhaps indicating longer-term stability in these measures.

A MANOVA detected a significant difference of a second order interaction, suggesting that the differences in the wrasse assemblage among locations and between habitats varied over time (Table 5.3). The CDA based on the site-time classification was used to illustrate this complex interaction. The first four canonical discriminants explained 65.1% of total variance. The ordination plots of the first two canonical discriminants (49.3%) illustrated the main trends in community structure in space and time, and the ordination plot indicates the main species responsible for these patterns (Fig. 5.8 and see also Appendix 5.4). Neither axis could clearly be attributed to either spatial or temporal changes alone, but the influence of locality was slightly greater than temporal variation, and the effects of habitat relatively minor. Most sites at Cattle Bay were usually aggregate at the upper right hand quarters of the plot at all times, while most sites at North Pioneer Bay tended to be aggregated at the center and lower right hand quarters of the plot. This region was characterised by a moderate abundance of most species. Most sites at South Pioneer Bay appeared to cluster at the left hand side of the plot, which suggests that the abundance of most species was lower than other localities.

Temporal variation at North Pioneer Bay appeared to be less than at South Pioneer and Cattle Bay (although not apparent from univariate graphs, Fig.'s 5.7 and 5.8). The time at which particular sites deviated most from their average structure varied among sites. At Cattle Bay, temporal deviation occurred at CE2 (when there were more *Hemigymnus melapterus* and *Cheilinus trilobatus*), CS4 (when *H. melanurus* was absent), CE5 (when there were more *H. melanurus* and *Stethojulis strigiventer*) and CE9 (when *S. strigiventer* and *Cheilinus fasciatus* were absent). Wrasses in both reef slope and edge habitats of North Pioneer Bay surveyed at time 7 were clearly distinct from the other times, because of the absence of *H. melanurus*. At South Pioneer Bay, three groups can be separated according to the abundance of *H. melanurus*. Low abundance was detected at time 5 and 9 from both habitats. High abundance were recorded at SE3, SS2, SE6 and SS7 while the rest were at moderate abundance. It was likely that temporal variation was mainly attributed to fluctuations in *H. melanurus*. However, there was also some variation attributable to other minor species in some habitats. For example, SS2 had more *S. strigiventer* than SE2, SE3 had more *Choerodon anchorago* than SS3, SS6 had more *Choerodon fasciatus* than SE6, and SE8 and SS10 had more *T. lunare* than SS8 and SE10.

Overall, there were no long-term trends in community structure, and those changes that were observed were short-term, small in magnitude and specific to species and locations.

## 5.5 DISCUSSION

### 5.5.1 ASSEMBLAGES STRUCTURE OF NEAR-SHORE REEFS

It has been argued that the processes/factors that determine spatial patterns in the structure of coral reef assemblages vary, depending on the spatial scale of interest (Williams 1991). On a large-scale, including and longitudinal and latitudinal gradients, and differences among reefs, larval supply and recruitment patterns in recruitment appear to be important (Williams, 1983b, Russ, 1984a, b; Doherty & Williams 1988; Williams 1991). However, since habitats also vary on these scales (Done 1982; Dinesen, 1983), the potential that habitat structure ultimately controls pattern through an interaction between recruitment patterns and habitat availability may be very important. Patterns of distribution of food and shelter, and associated patterns of habitat selection by fishes can be important in explaining differences in community structure among locations within reefs, and differences between habitat zones (Williams, 1991). It seems, therefore, that habitat-structure may be implicated at all these scales, although the processes it interacts with may differ. Clearly, integrated studies on fish-habitat relationships are necessary to provide hypotheses about the role of habitat in more general models of the processes influencing the distribution and abundance of coral reef fishes.

The objective of this study was to investigate the spatial pattern of fish-habitat relationships at three scales on near-shore fringing reefs. The large-scale pattern, or differences among the two islands, explained most of the variation in community structure. This may be part of a larger inshore-offshore gradient in community structure (as Magnetic Island is closer to shore) or a latitudinal gradient (as Magnetic Island is further south). The former is most likely as latitudinal gradients in the biota are stronger in this region. Also, fish species and habitat types characteristic of Magnetic island have also been recorded at Dunk Is, to the north of Orpheus (unpublished data).

The habitat structure of Magnetic Island was characterised by macroalgae, foliose coral, encrusting coral and Tabulate *Acropora*. In contrast, Orpheus Island was dominated by soft coral, dead coral, massive coral, coral debris, fire coral and branching coral. This pattern is likely to be the effect of longitudinal different (distance from land) rather than latitudinal different. The influence of latitudinal difference was less important because limited spatial scale and a bit board benthic categories used in this study. Bradbury *et al.* (1986) described the longitudinal patterns of coral reef benthic assemblage of the Central section of the GBR in terms of benthic lifeforms, but unfortunately, no effort was made to describe the near-shore reefs. Done (1982) described the longitudinal pattern of coral assemblages in the central section of the GBR where near-shore reef was classified as class III: communities of sheltered habitat and also termed as non-*Acropora* reefs. This was because the paucity of *Acropora,* and the proliferation of *Montipora* (CE), *Fungia* (CS), *Porites* (CM), *Goniopora* (CS), *Turbinaria* (CF) and *Galaxea* (CE). The distribution patterns of soft coral assemblages of inshore reef were described as low to moderate cover, when compared with hard coral, and never make a significant contribution on the reef (Dinesen, 1983). In contrast, this study found soft coral (SC) dominated the benthic community of some sites at Orpheus Island. For the flora component, cross-shelf distribution patterns for algae were briefly described only for mid- and outer shelf (Bradbury *et al.*, 1986). However, it was suspected to dominate on the near-shore reef. Vuki & Price (1994) mentioned that the macroalga, *Sargassum,* dominated the shallow water benthos on fringing coral reefs and is not important on mid- or outer shelf reefs. From this study it is clear that *Sargassum* is an important component of the near-shore reefs of Magnetic Island.

The most abundant wrasse species, *Halichoeres melanurus,* dominated both island locations. Despite this, the two assemblages were distinct. The wrasse assemblage of Magnetic Island was characterised by *Halichoeres dussumeri, H. biocellatus, Choerodon graphicus* and *Cheilinus trilobatus*, while that at Orpheus Island was characterised by *Labroides dimidiatus, Thalassoma lunare, Hemigymnus melapterus, Cheilinus fasciatus* and *H. chloropterus*. As with habitat, this difference was likely to represent part of a longitudinal gradient. Williams (1982) described that inshore wrasse assemblages were dominated by *Thalassoma lunare* and *Hemigymnus melapterus*. Williams & Hatcher (1983) found species richness and biomass of wrasse assemblage on near-shore reefs lower than that of mid- and outer reefs. However, the details at species level were not given. The latitudinal variation of fish assemblages do not appear to become apparent until much larger spatial scales (Williams, 1983a). Green (1996) described assemblage structure of wrasse at lizard Island (ca. 30 km from shore), northern Great Barrier Reef and recorded 64 species in which dominate by eight species. The results of Green (1996) may be regarded as offshore wrasse assemblage which appear to attain higher species richness than inshore reefs.

### 5.5.2 SPATIAL SCALES AND FISH-HABITAT RELATIONSHIPS

Regardless of whether the differences among islands are part of a gradient or not, the results of the Canonical Discriminant Analysis suggest a close association between fish assemblage changes and habitat changes on this scale. Some differences in overall community measures, such as species richness, diversity, evenness and overall abundance were also observed, but clearly these measures were less sensitive. The influence of habitat structure on fish assemblage is clearly shown by the nature of the relationship between habitat and fish, and also by a specific relationship between individual fish species and benthic lifeforms. As a family, wrasses tend to be associated with dead coral and soft corals (positive linear relationship), and are less associated with living coral and algae. However, at the species level, associations with all these habitats can be found, and these associations contribute to explaining large scale patterns. For example, association between *Halichoeres dussumeri and Choerodon graphicus* with macroalgae at Magnetic Island was detected. Overall, wrasses may indicate the effects of a wide range of disturbances effecting different components of the benthic habitat, not just changes in hard coral cover.

Larger-scale variation in fish-habitat relationships must be investigated further. In this study, a number of relationships between wrasse and habitat parameters were detected. In contrast, Green (1996) found no clear relationships between wrasse density or diversity and other habitat features such as living substratum cover or substratum diversity or complexity from Lizard Island, northern Great Barrier Reef. It is not known whether this represents a regional difference. Studies at other locations, including inshore reefs on the northern GBR (Jones, unpublished data), Phuket (Chapter 4) and Tuvalu (Jones & Kaly 1995) indicate a close association between labrids and habitat characteristics. Clearly, generalisations should be made with caution at this stage.

Variation among locations within reefs was also clearly very important, although the association between fishes and features of the habitat were less clear. There were differences in overall species richness and diversity etc. among locations, and considerable differences in community structure, particularly at Orpheus Island Wind and wave exposure may be responsible for some of this variation. All sites at Magnetic Island were exposed reefs, therefore, there was less variation among locations on this island. For Orpheus Island, three localities were exposed (Iris point, NE-reef, and N-Fantome) while the rest were sheltered. The different habitat structure for these three exposed localities was detected (CDA), and was due to the high abundance of soft coral (Fig. 5.4A) especially *Sarcophyton, Sinularia, Lobophytum* and *Nepthea* (personal observation). Dinesen (1983) described the composition of soft coral on exposed reefs, which is dominated by *Lemnaria, Sinularia, Lobophytum, Sarcophyton, Dendronephthya, Capnella* and *Nephthea.* The results for the wrasse assemblage appeared to follow the same trend, but were less distinct than those of habitat structure (Fig. 5.4B).

Differences among the two reef zones were low, both in terms of habitat structure and wrasse assemblages. This may be expected if habitat is important, but variation in habitat limited. The similarities among zones are not typical of reefs generally (Sakai *et al.*, 1987). The pattern on near-shore reefs may be explained on the basis of limited reef development, both in terms of the distance to the reef edge and depth (Done, 1982).

### 5.5.3 TEMPORAL VARIATION AT DIFFERENT SCALE

The temporal variation detected here mainly corresponded to seasonal or short-term variation, rather than differences among years. The demographic factors that are responsible for the pattern are likely to be recruitment and mortality. January appeared to be a summer recruitment period and August a winter recruitment period (Williams, 1983b). Seasonal patterns in mortality may also be important (Eckert, 1987).

Eckert (1987) estimated adult and juvenile mortality rates for wrasses, and suggested that species with low mortality rate require little recruitment to maintain abundance. Conversely, species with high mortality need high recruitment to maintain numbers. The mortality rates of several wrasses estimated by Eckert (1987) may be used to classify fish into three groups: low (<34%), moderate (34-66%) and high mortality (>66%). In this study, *Cheilinus chlorourus* and *Thalassoma lunare* can be considered as low mortality species, *Labrichthys unilineatus, Labroides dimidiatus* and *Hemigymnus melapterus* moderate and *Halichoeres melanurus, Coris batuensis* and *Stethojulis strigiventer* high. The seasonal variation of the whole assemblage should be contributed mainly from species of high abundance and high mortality species. Only two species, *H. melanurus* (50%) and *T. lunare* (10%) dominated in the sheltered reef of Orpheus Island. The temporal variation of total wrasse abundance, however, was very similar to that of *H. melanurus*, which indicates a significant contribution to the whole assemblage by this species. Other species e.g. *L. dimidiatus* and *H. melapterus* also displayed a similar temporal pattern to that of *H. melanurus*. The species concerned suggests that patterns of variation in mortality may explain the species responsible for temporal trends.

At an annual scale, all community parameters and the abundance of each species observed at the same summer recruitment periods (January) appeared to be similar. This indicated relatively stable assemblage of this area over a two-year period. This may indicate little disturbance from any source (both natural and anthropogenic sources) in this area. The temporal variation of benthic assemblage was not investigated because there was no sign of habitat structure changes during the study period. Seasonal variation of habitat structure can occur naturally due to macro-algae like *Sargassum* (Vuki & Price, 1994) which is not happen in the study area. For annual variation, only significant disturbances from various sources e.g. cyclone, *Acanthaster planci*, bleaching and extensive anchoring can cause dramatic changes in habitat structure, but this did not happen during the study period. The effects of nutrient enrichment and sedimentation may not be detected over a two-year time scale.

### 5.5.4 CONCLUSION

In conclusion, significant and deterministic relationships between habitat structure and wrasse assemblage were detected. This result indicates that the near-shore environment plays a key role in controlling the structure of coral reef benthic and wrasse assemblages. A variety of species-specific associations with particular features of the habitat appear to explain overall changes in community structure on the scales that changes in habitat structure are observed. The summation of these patterns also results in overall patterns between wrasse diversity and features of the habitat, with a greater number of species associated with dead coral and non hard coral substrata, compared with live coral or macroalgae. These relationships for wrasses might be used to indicate the status of coral reefs. It could be speculated that wrasse assemblages may reflect successional sequences, such as the stage of coral reef degradation when algae is being replaced by soft coral. The ability of different wrasses species to respond to particular features of the habitat make them a superior multi-disturbance indicator, compared with other fish families. The temporal stability of fish assemblages in the absence of changes to habitat structure is also a vital characteristic of a good group of indicator species. However, the patterns described in this study only represent correlations, not cause and effect. Tests of the causative role played by habitat are carried out in the next two chapters.

Table 5.1. Results of MANOVA showing spatial variation of A) habitat structure (16 categories) and B) wrasse assemblage (17 species) at 14 localities each with two habitats from central Great Barrier Reef. Type III SS&CP matrix, * significant at P < 0.05.

| | A) Habitat structure | | | B) Wrasse assemblage | | |
|---|---|---|---|---|---|---|
| Source | df | F | P | df | F | P |
| Locality, 1 | 208, 1716 | 5.59 | < 0.001 | 221, 1703 | 3.70 | < 0.001 |
| Habitat, 2 | 16, 120 | 8.91 | < 0.001 | 17, 119 | 5.96 | < 0.001 |
| 1 x 2 | 192, 1572 | 3.06 | < 0.001 | 204, 1560 | 2.09 | < 0.001 |

Table 5.2. Results of A) Canonical Correlation Analysis and B) Redundancy Analysis of the first three canonical axes showing the relationships between coral reef fish assemblages and habitat structure.

| | Canonical Correlation | % Variance |
|---|---|---|
| A) Canonical Correlation Analysis | | |
| Axis 1 | 0.95 | 72.07 |
| Axis 2 | 0.72 | 7.70 |
| Axis 3 | 0.65 | 5.29 |
| | Habitat structure | Wrasse assemblage |
| B) Redundancy Analysis | | |
| (% variance explained) | | |
| Explained by their own variable set | | |
| Axis 1 | 31.56 | 40.00 |
| Axis 2 | 10.59 | 8.26 |
| Axis 3 | 4.98 | 4.79 |
| Explained by opposite variable set | | |
| Axis 1 | 28.67 | 36.33 |
| Axis 2 | 5.44 | 4.25 |
| Axis 3 | 2.10 | 2.02 |

Table 5.3. Results of MANOVA showing temporal variation of wrasse assemblage (18 species) at three localities each with two habitats on Orpheus island. Type III SS&CP matrix, * significant at $P < 0.05$.

| Source | df | F | P |
|---|---|---|---|
| Time, 1 | 162, 2439 | 3.52 | < 0.001 |
| Locality, 2 | 36, 528 | 14.41 | < 0.001 |
| 1 x 2 | 288, 4448 | 2.49 | < 0.001 |
| Habitat, 3 | 18, 263 | 6.92 | < 0.001 |
| 1 x 3 | 162, 2439 | 1.89 | < 0.001 |
| 2 x 3 | 36, 528 | 5.36 | < 0.001 |
| 1 x 2 x 3 | 288, 4448 | 1.68 | < 0.001 |

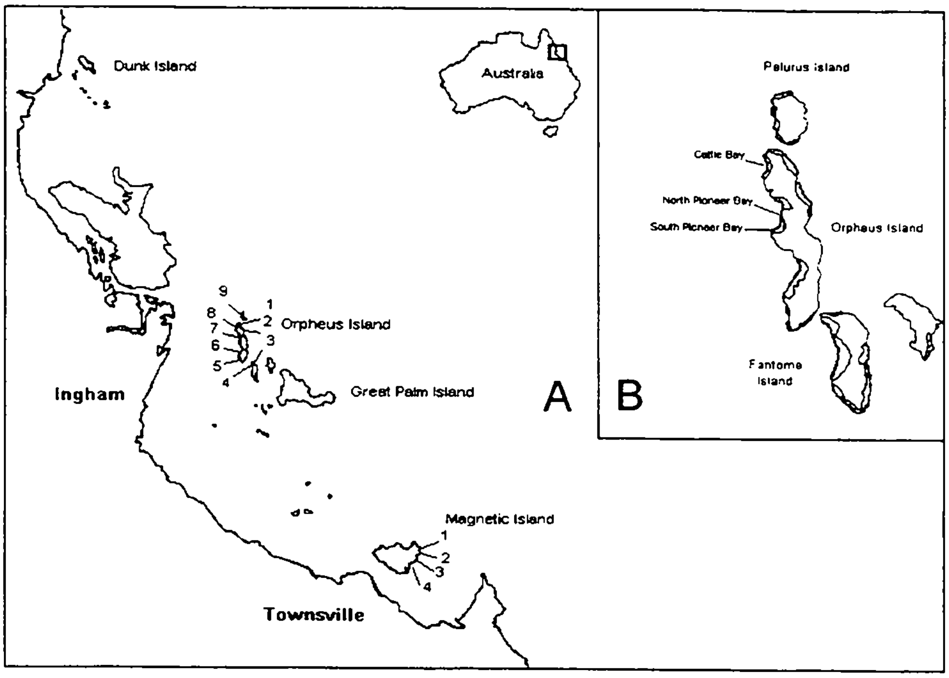
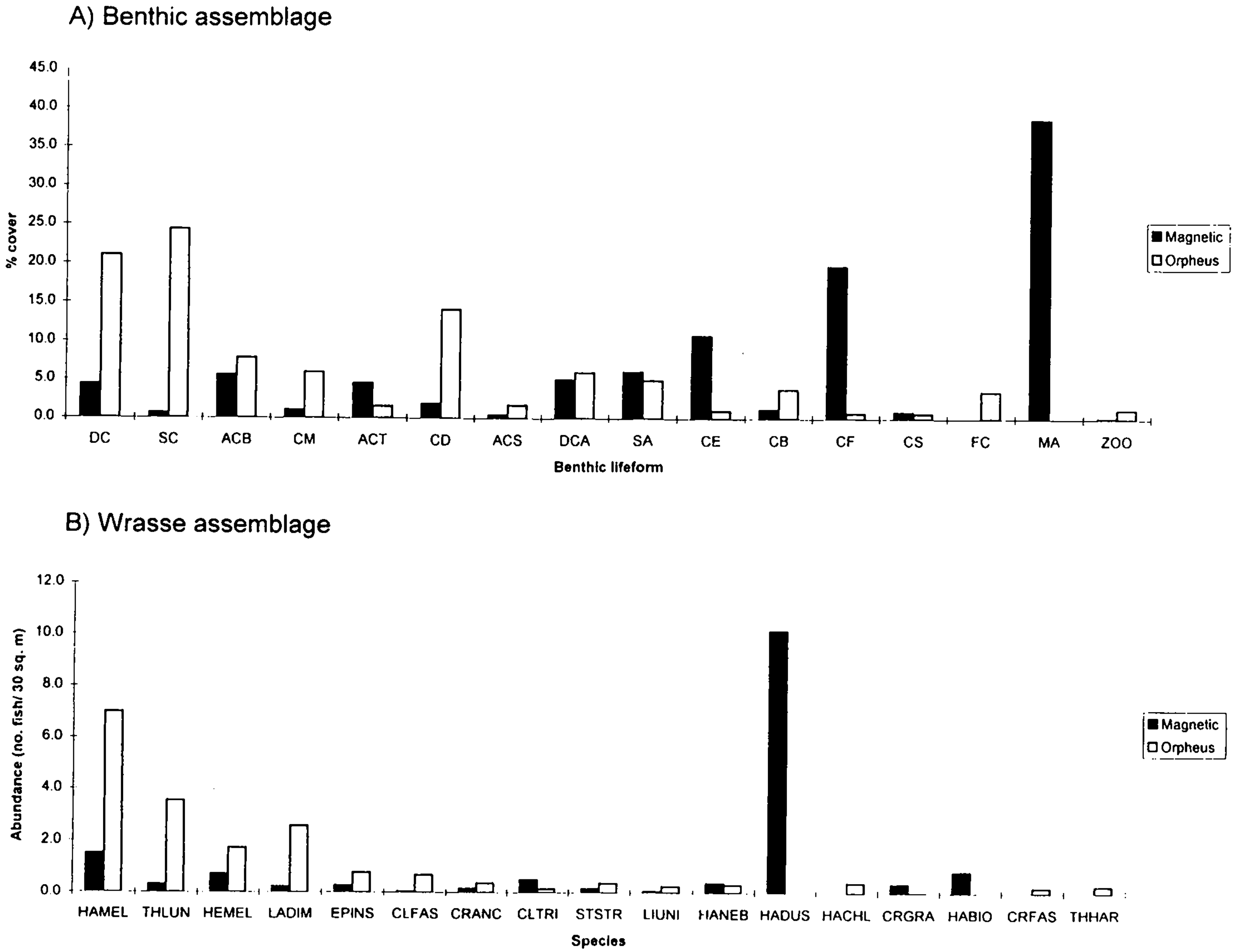


Fig. 5.1. Maps showing study area in the central Great Barrier Reef. A) 13 localities for spatial pattern at Magnetic Island (4) and Orpheus Island (9), and B) three locality at Orpheus Island for temporal pattern.


A) Benthic assemblage

B) Wrasse assemblage

Fig. 5.2. The different between Magnetic and Orpheus Islands in terms of A) Benthic assemblage composition and B) Wrasse assemblage composition (values were averaged across locality and habitat). Benthic lifeforms and wrasses are defined in Appendix 5.1 and 5.2 respectively.

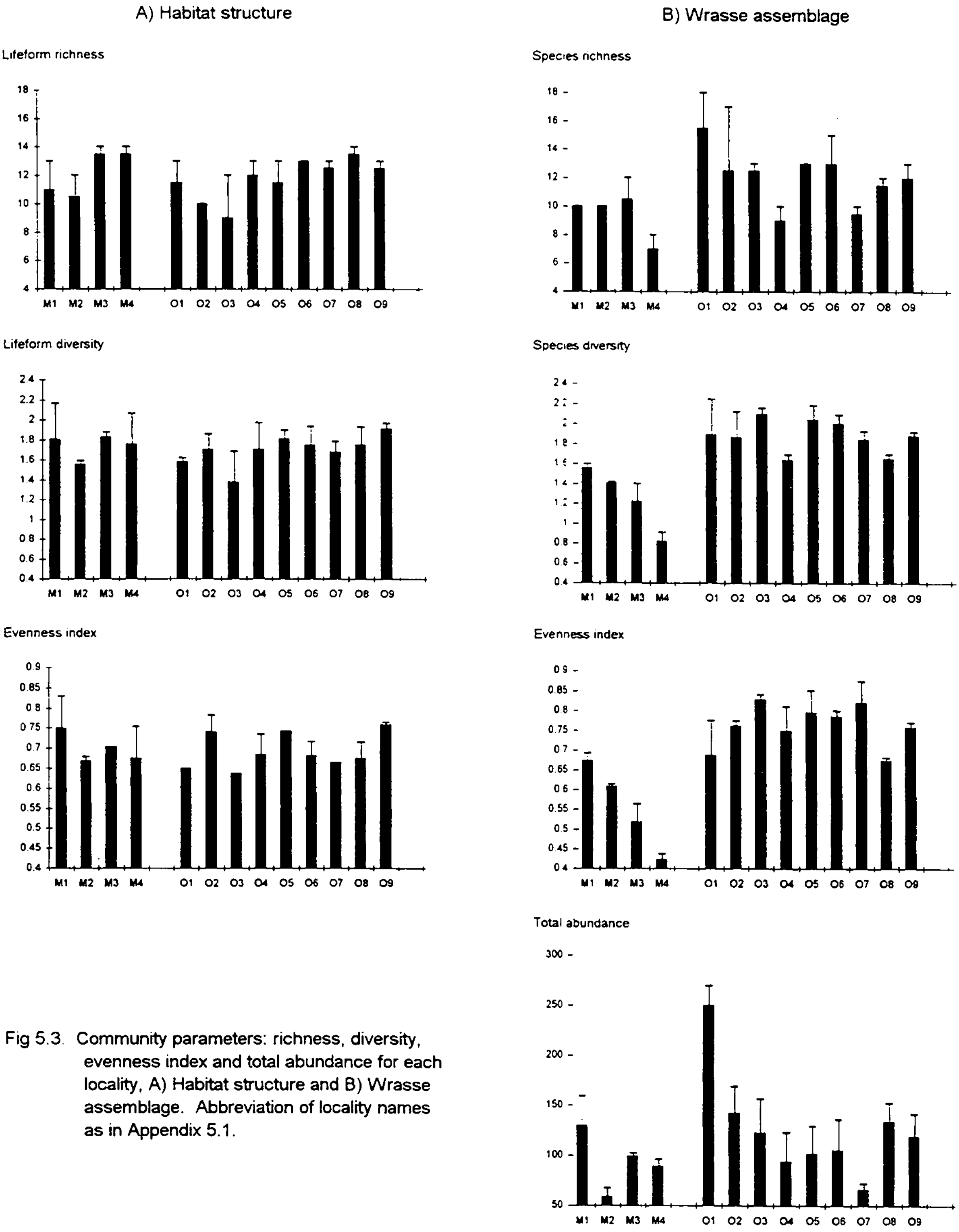


Fig 5.3. Community parameters: richness, diversity, evenness index and total abundance for each locality, A) Habitat structure and B) Wrasse assemblage. Abbreviation of locality names as in Appendix 5.1.

# Habitat structure

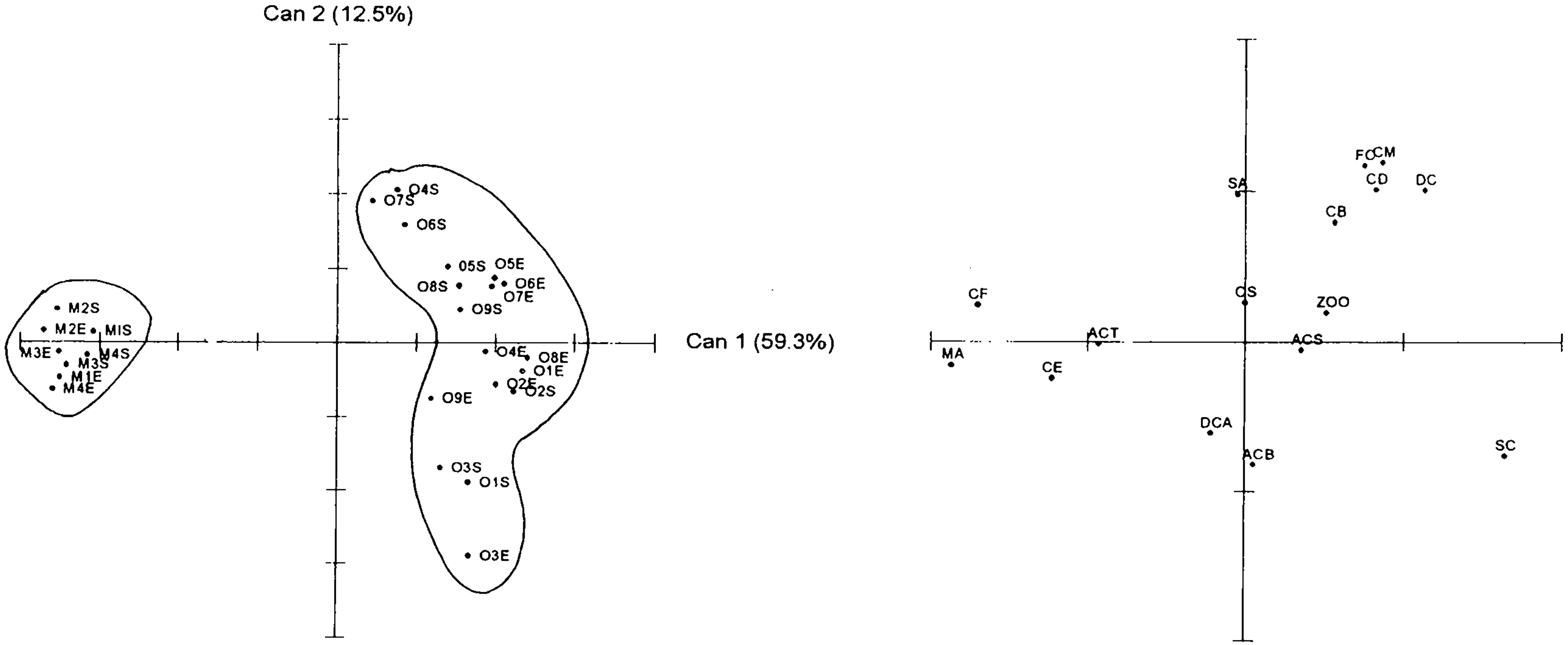


Fig. 5.4A. Ordination plots of CDA showing spatial pattern from habitat structure on the central Great Barrier Reef. Site name codes, and abbreviations of lifeforms are presented in Appendix 5.1.

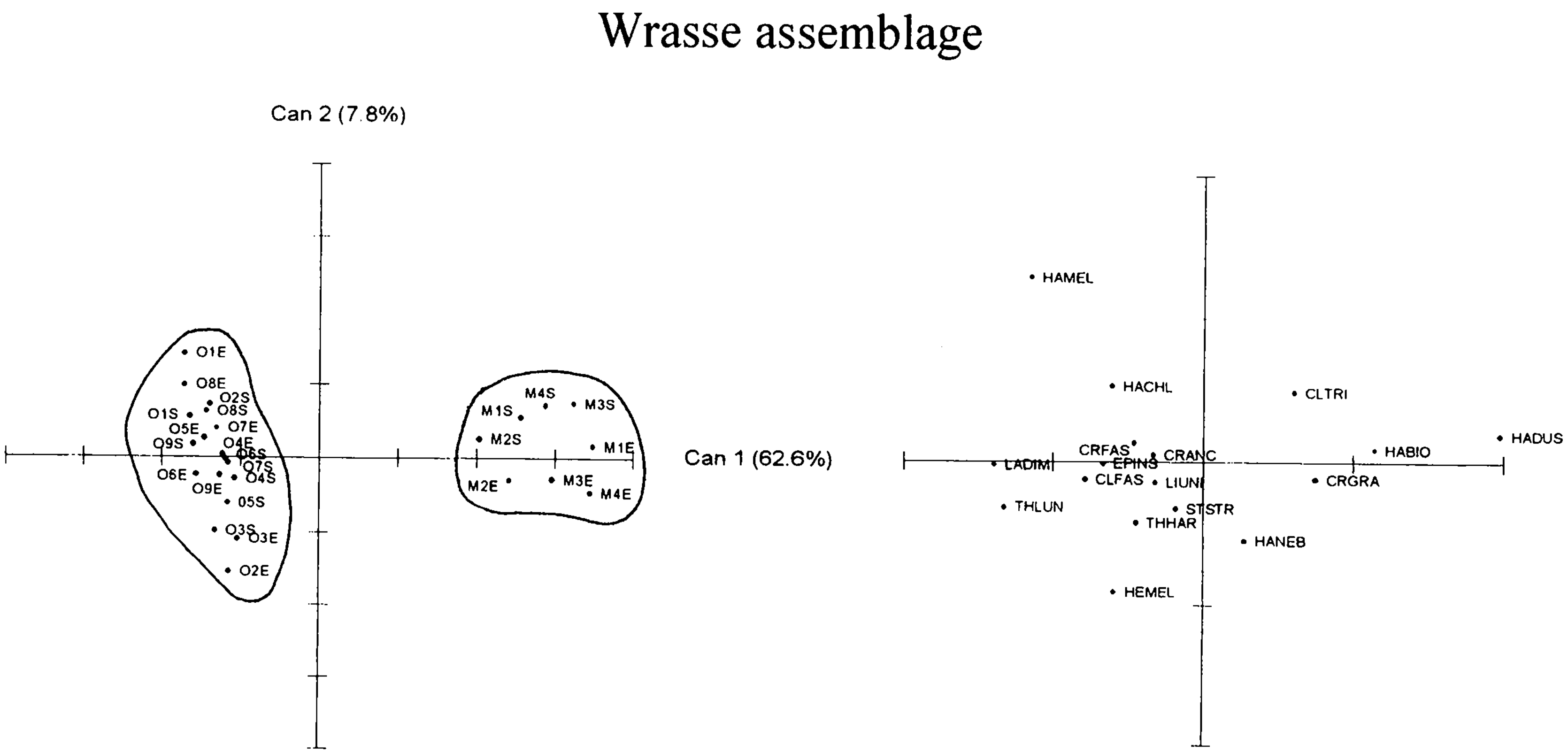


Fig. 5.4B. Ordination plots of CDA showing spatial pattern from wrasse assemblage on the central Great Barrier Reef. Site name codes, and abbreviations of wrasse are presented in Appendix 5.2.

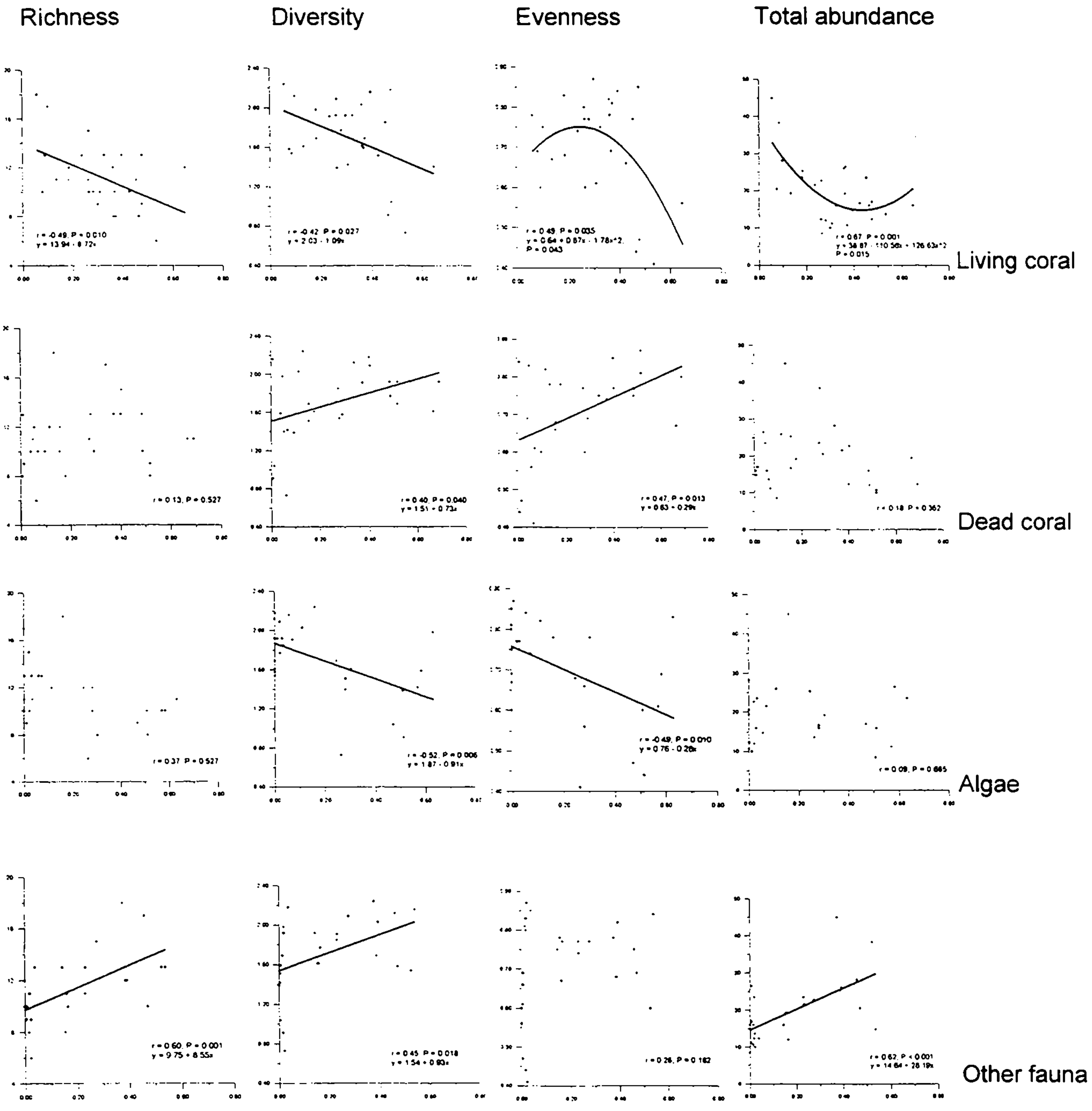


Fig. 5.5. Relationships between area cover of four major benthic lifeforms and four community parameters of wrasse assemblage. Regression lines are shown only for significant model.

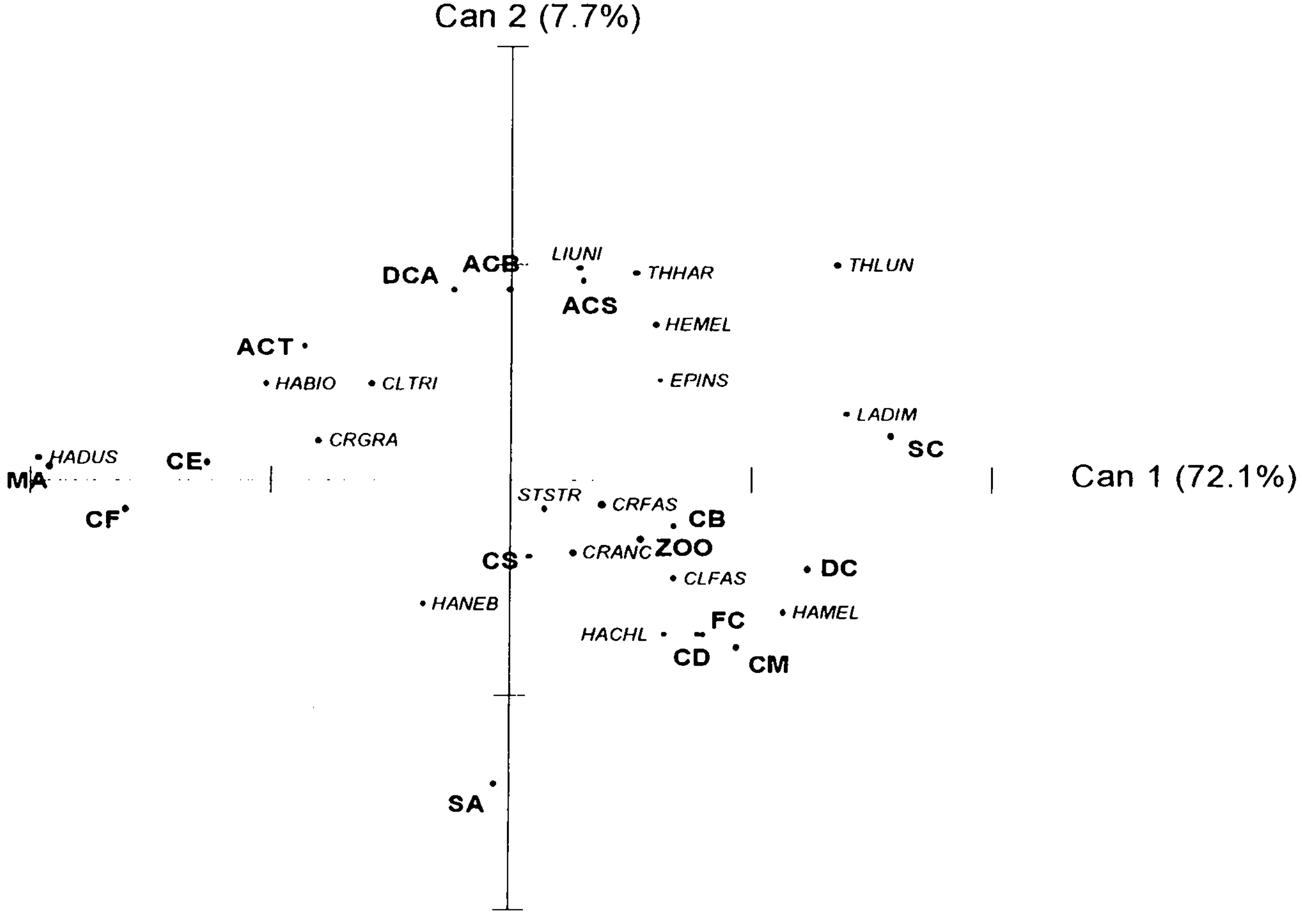


Fig. 5.6. Ordination plot of CCA showing trends in relationships between habitat structure (capital bold letter) and wrasse (italic letter). Abbreviations of benthic lifeforms are as presented in Appendix 5.1 and for wrasses as in Appendix 5.2.

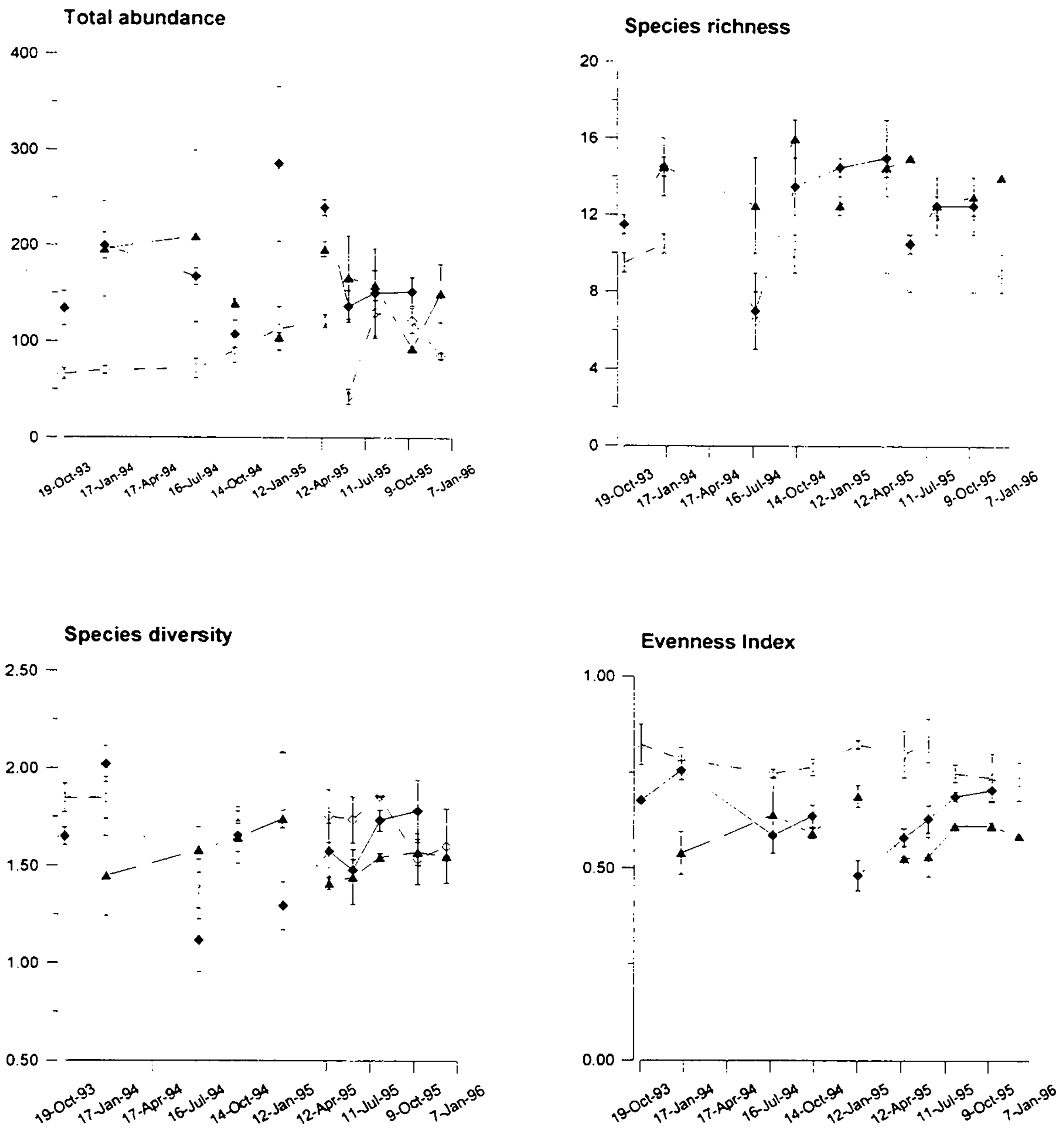


Fig. 5.7. Temporal patterns of four community parameters for wrasse assemblage from three localities at Orpheus Island. Mean between habitats are presented. ▲ = Cattle Bay, ☐ = North Pioneer Bay, ◆ = South Pioneer Bay.

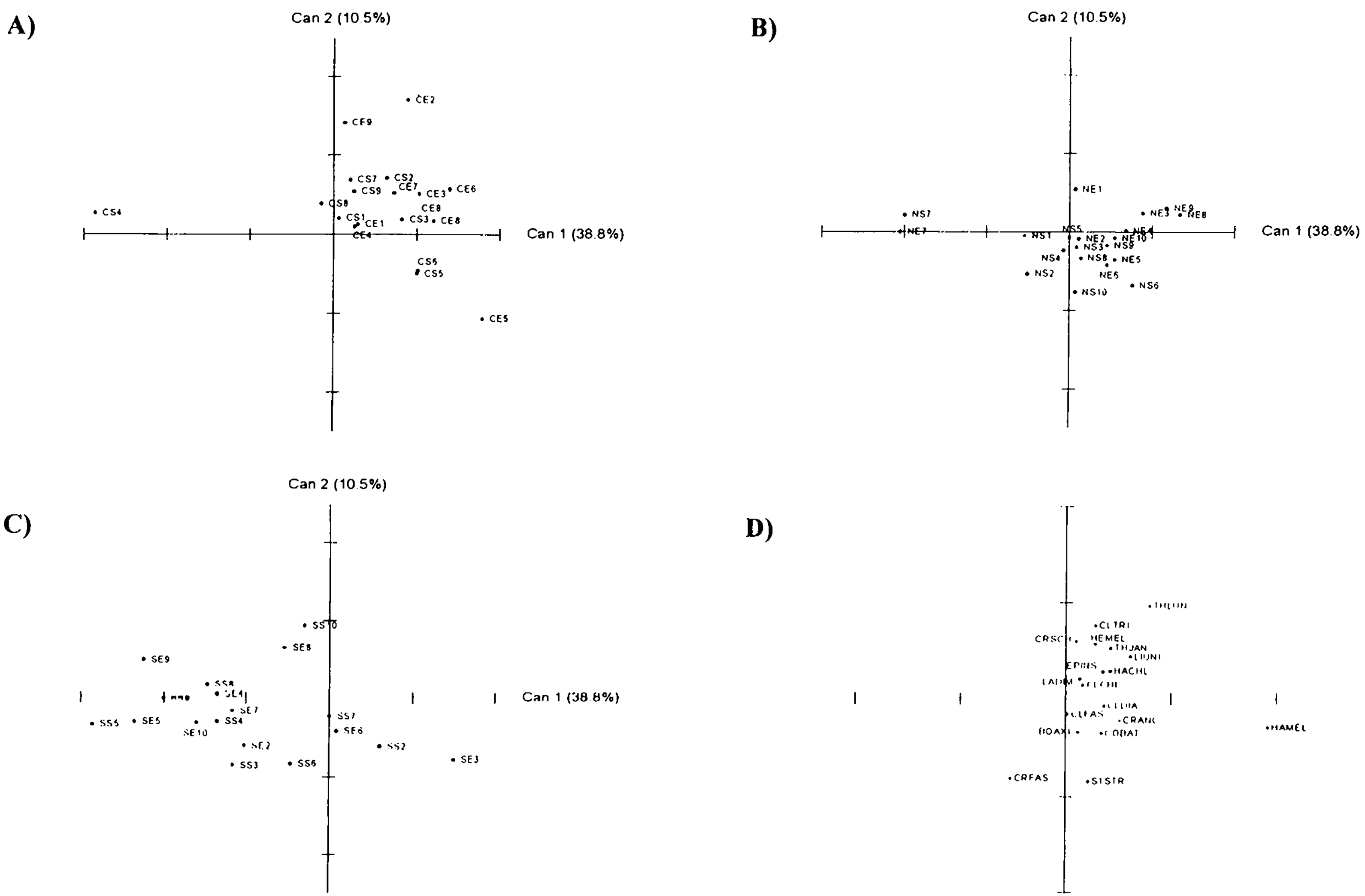


Fig. 5.8. Ordination plots of CDA showing temporal variation of wrasse assemblage at Orpheus Island. Separate plot for each locality, A = Cattle Bay, B = North Pioneer Bay, C = South Pioneer Bay and D) = h-plot. Site code name was represented by a three letters and nimbers. The first letter represents locality (C = Cattle Bay, N = North Pioneer Bay, and S = South Pioneer Bay), the second letter represents habitat (S = Slope and E = Edge) while the number represents data collecting times (1 - 10). Abbreviations for species of wrasse are as shown in Appendix 5.2.

# CHAPTER 6

# EXPERIMENTAL EVALUATION OF THE EFFECT OF LOCAL HABITAT DEGRADATION ON CORAL REEF FISH ASSEMBLAGES AT TWO GEOGRAPHIC LOCATIONS

## 6.1 SUMMARY

The effect of mechanical disturbance of the coral habitat on reef fish assemblages was experimentally investigated at two spatial scales, regional and local. At the regional level, near-shore and highly impacted reefs at Phuket (Thailand) were compared with inshore, relative non-impacted reefs near the central Great Barrier Reef (Australia). Local variation was assessed by repeating experiments at two sites within each region. The structure of fish assemblages differed between the two regions in relation to biogeographic differences in the species pool. Phuket was dominated by both pomacentrids and labrids, while pomacentrids dominated at central GBR. Experimental manipulations involved a reduction in the cover of live hard coral (GBR and Phuket) and a reduction in live soft coral (GBR only), with appropriate controls. The response of fish to habitat disturbance varied regionally and locally, and depended on which fish taxa was examined and whether hard coral or soft coral was disturbed. In general, most fish taxa responded negatively to a reduction in living coral cover, with decreased diversity, species richness, and reduced abundance of coral-associated species. In contrast, removing soft coral appeared to have a positive effect on the abundance of many fish species, perhaps because it resulted in an increase in habitat complexity. The magnitude of the influence of habitat degradation appeared to differ between the two regions, with Phuket exhibiting greater extent of impacts with less tendency for recovery. In contrast, impacts on the central GBR were slight and recovery was rapid. The experimental manipulation, by reducing components of the habitat, illustrated that habitat structure and resource availability may have a strong impact on the structure of reef fish assemblages at widely separated locations.

## 6.2 INTRODUCTION

Benthic coral-reef assemblages represent an important component of habitat structure for fishes associated with coral reefs, providing food and shelter (Jones 1991). Natural disturbance has been shown to have a major influence on the structure of these benthic assemblages (Highsmith, 1980; Sano *et al.*, 1984, 1987; Brown *et al.*, 1993), and in doing so, may influence the availability of resources for fishes. Increasingly, additional anthropogenic factors have contributed to changes in benthic community structure, reducing the cover of live corals and altering habitat complexity (e.g. Pastorok & Bilyard, 1985; Tomascik & Sander, 1987; Sorokin, 1993). The effects of these kinds of disturbances on fish communities are poorly understood, whether it be general community level responses (e.g. species richness) or effects on individual species and more subtle effects on community structure. While some effects of large-scale disturbance processes have been documented at some locations (Kaufman, 1983; Lassig, 1983; Wellington & Victor, 1985; Williams, 1986; Sano *et al.*, 1987), studies on other taxa at other locations have demonstrated inconsistent effects (Tabb & Jones, 1962). The importance of disturbance may vary regionally and locally, but there have been few experimental studies on which to base any generalisations.

Habitat structure can be regarded as an important factor influencing the structure of coral reef fish assemblage, both at settlement (Sale *et al.*, 1984; Eckert, 1984; Jones, 1991; Green, 1994) and during the post-recruitment phase of the life cycle (Jones, 1991). It appears that the effect changes in habitat structure have on fishes varies among species or group of fishes and/or locations (Roberts & Ormond, 1987; McCormick, 1994; McClanahan, 1994; Green, 1996). Habitat changes necessarily involve a decrease in some substrata and a corresponding increase in others. Hence if particular habitat structures represent limiting resources at some stage in the life cycle, both negative and positive responses might be expected. However, most studies have focused on particular groups, and no general pattern has emerged (see Sano *et al.*, 1984; 1987; Wellington & Victor, 1985).

The scale and type of disturbance may have different influences on both habitat structure and associated fish assemblages. A catastrophic level of disturbance, such as that from severe storms, may have a direct impact on physical structure of the habitat over broad areas. The relative importance of these factors on associated reef fishes is unknown (Kaufman, 1983; Lassig, 1983).

Impacts from biological agents such as *Acanthaster* change the biological composition of benthic assemblage, but in the short-term do not mechanically damage the physical structure. It has been argued that only fish that directly utilise live coral for food (e.g. many butterflyfishes) will be impacted by such disturbances (Sano *et al.*, 1984; Williams, 1986) while other groups may not show any response. However, in the longer term, *Acanthaster* infestation can cause physical changes as a result of secondary agents such as borers, grazers and also wave and wind action. These physical changes can have a direct impact on a range of resident fishes such as pomacentrids and labrids, due to changes in topographic complexity and the availability of shelter (Sano *et al.*, 1987).

The structural changes of coral reef fish assemblages associated with more localised mechanical damage, due to moderate storms, localised outbreaks or anchor damage have not been studied. However, these kinds of disturbances may be more frequent and widespread in coral reef communities. Effects may not be detected from field observations alone, due to the patchy nature of local reef habitats. While effects may be rapid at such scales, recovery may be equally rapid, but this is not known.

In all cases, it is necessary to identify whether the impacts on fish population or assemblages come directly from disturbance or via changes in habitat structure (Jones, 1991). The two alternatives can be distinguished by experimental manipulations of the habitat. Experiments can be conducted to determine the specific effects of disturbance to different substrata, the magnitude of the disturbance effect necessary to induce impacts and the nature of species-specific responses among the fish fauna. To date, most of the published experimental studies have been limited to relatively small and isolated areas (e.g. Sano *et al.*, 1984, 1987; Caley, 1993) which limit their applicability to patches on large and continuous reef systems (Doherty, 1991). By repeating experiments in different habitats and at different locations, the degree to which responses are of general significance can be determined. Processes that are important, regardless of geographic location and the species pool, will also be of considerable significance. The importance of additional disturbance may vary depending on the background status of the reef. It would be predicted that areas already heavily impacted may be resilient to further impacts, if most of the effects have already occurred.

In this study, experimental manipulations were carried out to investigate the influence of small scale habitat degradation (reduction in coral cover) on the structure of coral reef fish assemblage on contiguous reef. Similar experiments were carried out at two geographic locations, the highly impacted reefs of Phuket (Thailand) and the relatively unimpacted fringing reefs of the inshore islands of the central GBR (Australia), and at two locations within each of these regions. The aim was to detect common responses at both population and community levels, for a variety of taxa.

## 6.3 MATERIALS AND METHODS

### 6.3.1 STUDY AREA

Experiments were conducted at two geographical locations. Phuket (Thailand) and central Great Barrier Reef (Australia). At Phuket, field experiments were carried out at two localities at Maiton Island (7° 45′ N, 98° 29′ E ) 8 km south east of Phuket (Fig 6.1A). At central GBR, field experiments were repeated at two localities on Orpheus Island (18° 35′ S, 146° 29′ E), 16 km offshore from Ingham (Fig. 6.1B).

At Maiton Island, there is reef development around most of the island's perimeter. Benthic reef assemblages on the east side have developed on the sandy bottom, while on the west side they have developed on the rocky shore. The degree of reef development on the east side is greater than on the west side, which can be attributed to the influence of monsoon. The southwest monsoon has more effect than the northeast monsoon. During the last 10 years, almost all reef benthic assemblages have been destroyed, particularly by storms (Chansang, Phuket Marine Biological Center, *per. comm.*). At present, there is no indication of further impacts, but neither has there been significant recovery. The establishment of a local resort in the early 1990's, has had potentially positive and negative effects. The resort operation has provided some degree of protection from illegal fishing activities. However, there may also be impacts from the resort itself, particularly nutrient enrichment.

The details of benthic-reef assemblage and coral reef fish were described in Chapter 4. Two localities, one on the northeast and one on the southeast were selected as study sites. The northeast site was situated in front of the Maiton Holiday Resort where recent reef benthic assemblages are dominated by coral debris (small dead coral fragment). At the southeast site, reef benthic assemblages are dominated by sand and dead coral, which may indicate the influence of storms during the past few years.

At Orpheus Island (GBR), coral reef development occurs around the island, with extensive reef platforms and clear water on the windward (eastern) side and extensive sand flats or short rocky shores and turbid water on the leeward side. Most of the land and surrounding waters are part of a marine park, so impacts from human activities are few. Inshore, leeward reefs are also relatively protected from storm activities. The experiments were carried out at Pioneer Bay and Cattle Bay, both on the western side of the island.

## 6.3.2 EXPERIMENTAL DESIGN

### 6.3.2.1 Experimental unit

A 5x5 $m^2$ quadrat was used as the basic experimental unit. Three replicates were used for each treatment at Maiton Island while four replicates were employed at Orpheus Island. At Maiton Island, experimental units were set up at the depth of 4 to 5 m below mean sea level, near the lower limit of the coral zone on the middle of the reef slope. At Orpheus Island, experimental units were set up at 3 to 4 m below mean sea level or at the reef edge. The distance between plots varied between 5 to 20 m. The criteria for selection of experimental plots was based on stratified random sampling, with areas of moderate to high cover of living coral or soft coral selected. Benthic area covers of each experimental unit were assessed in the field prior to treatment allocation by using a 1 $m^2$ quadrat, using the intersections of a grid with 25 x 25 $cm^2$ spacing to estimate cover.

### 6.3.2.2 Treatments

**Experiment 1: Maiton Island, Phuket**. Three treatments were established at each location to examine whether a reduction in living hard coral cover influenced the fish assemblage, and whether experimental coral reduction result in communities naturally low in coral. It was predicted that if habitat was important, disturbed patches would diverge in community structure away from the structure of undisturbed patches, and converge on those with a pre-existing low coral cover. The three treatments were as follows (with three replicates per treatment):

Treatment 1: Living coral control (LC). Areas of approximately 20% coral cover that were not subject to disturbance.

Treatment 2: Living coral disturbance (LD). Areas similar to the control, but with coral cover reduced to approximately 5% cover, by translocation of living coral from experimental unit to at least 50 m away.

Treatment 3: Dead coral area (DC). Areas in a pre-existing disturbed situation, with coral cover of approximately 5%.

**Experiment 2: Orpheus Island, GBR.** A set of treatments almost identical to the Phuket experiment were established at each of the two sites, but with four replicates per treatment. The initial hard coral cover here was 50% (compared with only 20% at Phuket). Two additional treatments were established at the Orpheus Island locations to test the hypothesis that disturbance to soft coral would have a major influence on the structure of fish communities. The additional treatments were:

Treatment 4: Soft coral control (SC). Areas of soft coral cover ranging between 70 -100 %. The soft corals that dominate in the study area at both localities on Orpheus Island were bushy *Sinularia* spp. and some *Nepthya* spp.

Treatment 5: Soft coral disturbance (SD). Areas of high soft coral manipulated by removing approximately 90% of the existing soft coral.

It was predicted that fish community structure would diverge from controls in areas that were disturbed, and community structure would come to resemble unmanipulated areas of low coral cover.

### 6.3.3 DATA COLLECTION

Experiment 1 was conducted from April 1994 to March 1995, and Experiment 2 from October 1994 to January 1996. The fish assemblage was estimated by using a visual census technique, taking 10 to 15 minutes to monitor each plot. All of fish resident in or using all 5 x 5m plots were identified and recorded. However, cryptic and small species were excluded from the study, e.g. Gobiidae and Bleniidae. The fish assemblage data at Maiton Island were collected five times: 1) before habitat manipulation , 2) one week after manipulation, 3) to 5) were four, seven, and 10 months after manipulation. However, at time 3 (four months after manipulation) data were collected only at Northeast reef because of bad weather. For Orpheus Island, data were collected 10 times: 1) two weeks before habitat manipulation, 2) two weeks after manipulation, 3 to 5) every month and 6) to 10) every two months

### 6.3.4 DATA ANALYSIS

Three community parameters: species richness, species diversity, and overall abundance, of each experimental unit were analysed. Data from each region were considered separately. The diversity index used was the Shannon-Wiener diversity index (H') (Pielou, 1974). Separate analyses were carried out for all fish species, which were then divided into three groups: Labridae and Pomacentridae, and other major families. However, other major family groups were not considered for the experiment at Maiton Island due to their very low abundance and occurrence.

Analyses of Variance (ANOVA) were carried out to test hypotheses about the effects of hard coral and soft coral removal, with separate analyses for each region due to the different taxonomic composition. Two analyses were carried out for each locality, one using baseline data (before manipulation - 1 factor ANOVA) to examine pre-existing patterns, and one after manipulation, to examine treatment effects (2 factor ANOVA: Factors Treatment, Time). A multi-factorial ANOVA was not used because of lack of power. *A priori* tests was carried out for species richness and overall abundance to satisfy the assumption of homogeneity of variances. Only overall abundance data was square-root transformed. Student-Newman-Keuls (SNK) tests was used as *a posteriori* tests to identify the source of significant differences from main effects.

Canonical Discriminant Analysis (CDA) was used to test for a significant difference among treatments at each locality at each time, based on the distribution and abundance of fish in the experimental unit. Data reduction was carried out before performing the CDA by only using species present in more than 10% of total plots and times. Three groups/families: Labridae, Pomacentridae, and other major families, were analysed separately (except for experiment 1 at Maiton Island, for which other families were excluded). *A priori* tests were carried out and square-root transformations applied to satisfy the assumption of a multivariate normal distribution. This transformation was also used to reduce the influence of dominant species on the overall analysis.

Data from each of the two experiments were analysed separately, but both localities within each region and all sampling times were analysed together. The CDA was performed on square-root transformed and a centred data matrix. The analyses were conducted by Proc CANDISC in SAS (SAS Institute Inc., 1990). Species responsible for the discrimination were considered from the total canonical structure (correlation), because fish were considered to respond to the treatments rather than to have influenced (predict) them. The angular interpretation therefore was used to describe the ordination plot.

The selection of display canonical discriminant (variate) was based on the treatment at time 1, where the position of controls (LC1 and SC1) should be close to habitat degradation treatments (LD1 and SD1) while dead coral area may or may not be at the same position. To overcome the problem of overcrowding and because results may differ between localities, ordination plots were separately generated for each locality.

## 6.4 RESULTS

### 6.4.1 REGIONAL DIFFERENCES IN COMMUNITY STRUCTURE

There were considerable differences in community structure between Maiton and Orpheus Island, although the overall number of species recorded in the experimental treatments was similar. At Maiton Island, a total of 53 species (Appendix 6.1) from three main groups were observed: Pomacentridae (18), Labridae (24), and other major families - Chaetodontidae, Nemipteridae, Siganidae, Mullidae (11), with a total of 2,842 individuals recorded from the three treatments at northeast and southeast reef during five times between April 1994 to March 1995. Pomacentridae (53.5%) and Labridae (44.6%) dominated in the experimental area while other families (1.9%) could be considered as very rare. In terms of occurrence, seven pomacentrids and 14 labrids occurred at more than 10% of all sites & times (Appendix 6.2A). The dominant species were: pomacentrids - *Pomacentrus chrysurus, P. adelus, P. similis, P. moluccensis, Neopomacentrus azysron, Dascyllus aruanus* and *Chromis weberi*; labrids - *Thalassoma lunare, Halichoeres hortulanus, H. timorensis, H. vrolikii, H. marginatus, Coris batuensis, Stethojulis interrupta, H. scapularis, S. trilineatus, C. pictoides, H. argus, H. nebulosus, Cheilinus chlororus* and *Labrichthys unilineatus*.

At Orpheus Island, a total of 59 species (Appendix 6.1) from three groups; Pomacentridae (25), Labridae (20), and other major families (15), with a total of 30,867 individuals were recorded from the five treatments at Pioneer and Cattle Bays during the 10 sampling times between October 1994 to December 1995. Fish assemblages were dominated by pomacentrids (79.1%), but labrids were well represented (15.0%)

and other families were less common (5.9%). In terms of occurrence, 15 pomacentrids, 10 labrids, and four other families presented more than 10% of all sites and times (Appendix 6.2B). Common species were: pomacentrids - *Pomacentrus adelus, P. wardi, P. moluccensis, Neoglyphidodon melas, Neopomacentrus azysron, Amblyglyphidodon curacao, Acanthochromis polyacanthus, Abudefduf bengalensis, Dischistodus melanotus, P. alexanderae, A. sexfasciatus, Hemiglyphidodon plagiometopon, Chrysiptera rollandi, Chromis viridis* and *P. chrysurus*; labrids - *Halichoeres melanurus, Thalassoma lunare, Stethojulis strigiventer, Labroides dimidiatus, Hemigymnus melapterus, H. chloropterus, Epibulus insidiator, Cheilinus chlororus, Choerodon anchorago* and *Labrichthys unilineatus*; other families - *Chaetodon aureofasciatus, Scarus* spp., *Siganus doliatus* and *Scolopsis bilineatus*. There was only 21.5% (pomacentrids 26.5%, labrids 22.2% and other families 13.1%) overlap in species composition between the two regions, reflecting the different biogeographic species pool.

### 6.4.2 IMPACT OF DISTURBANCE ON COMMUNITY PARAMETERS

**Experiment 1:** Prior to experimental manipulations, there were few significant differences among the live coral and planned disturbance treatments, although fortuitously, at the NE-Reef site there was a significantly greater abundance of labrids and pomacentrids (and all families combined) in the live coral treatment at the start of the experiment (Table 6.1). After the experimental manipulations. major differences among the three different treatments at Maiton Island were detected, but the specific effects differed for the two localities (Table 6.1). The live coral disturbance treatment had a major impact at the NE-Reef location on most comparisons testing for effects on diversity, species richness and overall abundance (Table 6.1). At the SE-Reef location, the disturbance effects did not appear until the last sampling date. Most of the ANOVA results, however, failed to detect significant interactions between treatment and time (suggesting that the influence of treatments was consistent through time). At SE-Reef, most of the effects were due to differences between the dead coral area and the other treatments. The significant differences among times (natural variation) were detected only for abundance estimates for all fish species combined and pomacentrids at SE-Reef. To interpret particular treatment effects at each location, all variables were subject to an SNK tests (Table 6.1) and the magnitude of effects was interpreted graphically (Fig's 6.2-6.4).

In terms of species diversity, there was a significant effect of disturbance detected at NE-Reef, where coral disturbance caused overall species diversity to decrease (Fig. 6.2). Pomacentrids were almost entirely responsible for this effect. The effects of disturbance to labrid diversity at NE-Reef were small (Fig. 6.2). Overall, disturbed reefs show a reduction in diversity, which tended toward dead coral control levels. At SE-Reef, a substantial negative effect of living coral degradation could be seen from all three data sets, with experimentally disturbed sites undergoing a decline, relative to control sites (Fig. 6.2).

The effects of disturbance on species richness were similar to species diversity with most groups examined showing a decline on experimentally disturbed reefs, relative to the live coral control (Fig. 6.3). Species richness was extremely low on the dead coral control plots at all times. At the SE-reef site, species richness on disturbed reefs converged on that for dead coral controls by the end of the experiment.

A significant effect of disturbance on total fish abundance was apparent at both locations (Fig. 6.4). At NE-Reef, the abundance of fish on disturbed reefs declined to densities below that of the dead coral control. At SE-Reef, the abundance of fish on the disturbance reefs was higher than the other treatments prior to the experiment (see Table 6.1, ANOVA failed to detect significant treatment effect). At SE-Reef, significant differences were detected after manipulation but they were due to low densities on the dead coral controls. The effects of disturbance at this site appear to be primarily due to the labrids (Fig. 6.4).

**Experiment 2:** At Orpheus Island, the reduction in live hard coral cover also had a major influence on the community parameters (Table 6.2). Specific effects differed for the two localities, but generally the family Labridae showed no significant effects, while the Pomacentridae, "Other" fish families and the "Overall" community exhibited statistically significant treatment effects. As at Maiton Island, due to chance there were some significant differences prior among treatments prior to the experiment, but this was mainly restricted to the "Other" category at Cattle Bay (Table 6.2). After manipulation, the influence of time (temporal variation) were detected for most parameters of each group of fish. However, the effects of treatment were consistent across time due to non-significant differences of the "time x treatment" interaction.

The effects of disturbing hard coral on fish species diversity of all species combined was primarily due to effects on the pomacentrids (Fig. 6.5). At both localities, species diversity in disturbance treatments was consistently lower than in the live coral controls (Fig. 6.5). The effects on other families were less consistent through time, and some effects may have been pre-existing.

The results of species richness (Fig. 6.6) and total abundance (Fig. 6.7) were similar to that for species diversity, as negative effects from the coral disturbance were detected for all taxa combined and pomacentrids. Total abundance of labrids and other families also showed a similar patterns, suggesting a potential negative effect of the disturbance to hard coral (Fig. 6.7).

Disturbance to soft corals at Orpheus Island had significant effects on fish assemblages, but the effects were opposite to the hard coral disturbance (Table 6.3). That is, where effects on diversity, species richness and total abundance were detected, these increased on disturbed plots, compared with soft coral controls. In general, there were more effects detected at Pioneer Bay (Table 6.3). There were few significant differences among treatments before manipulation. There was temporal variation detected after habitat manipulation that were unrelated to the disturbance, particularly at Cattle Bay. However, the effects from treatments (if detected) were usually consistent across time (no significant effect from time and treatment interaction).

The effects of disturbance to soft coral areas on species diversity were detected on all species combined at both localities and labrids and pomacentrids at Pioneer Bay (treatmentXtime interaction also indicates the treatment induced a different temporal pattern) (Table 6.3). In general, overall fish, labrids and pomacentrids showed a similar trend in that species diversity in soft coral degradation higher than soft coral control for five to six months and appeared to be the same after that (Fig. 6.8). For other families, only short term effects of disturbance to soft coral degradation can be seen after manipulation. as species diversity increased sharply during the first three months and then dropped to equal or lower than the soft coral control (Fig. 6.8).

For species richness, significant differences between treatments after manipulation were detected only for all species combined and labrids at Pioneer Bay (Table 6.3). The species richness of both groups in the disturbance treatment at Pioneer Bay was higher than soft coral controls, but similar to the dead coral controls (Fig. 6.9). However, there was no effect after 9 months (time 7). There were no significant effects detected for pomacentrids and other families. However, there was only a short term effect detected on species richness of other families at Cattle Bay (Fig. 6.9).

An influence of disturbance to soft corals on total abundance was detected for all taxonomic groups, but was locality specific (Table 6.3). Significant differences were detected after manipulation for all species combined at Pioneer Bay, where abundance in the soft coral disturbance treatments was consistently higher than both controls (Fig. 6.10). A similar result was also detected for labrids and pomacentrids (Fig. 6.10). A significant difference among treatments was detected after manipulation for other families at Cattle Bay, but this was the influence of the dead coral control, not the soft coral manipulation (Fig. 6.10).

### 6.4.3 IMPACT OF DISTURBANCE ON SPECIES COMPOSITION AND COMMUNITY STRUCTURE

The ordination plots from CDA (results not shown) suggested that the significant differences for other major families for both experiments at Orpheus Island come from some temporal variation while there is no or less influence from treatment different. Respond of fish assemblages to habitat manipulation, therefore, were considered from labrid and pomacentrid assemblages.

Canonical discriminant analysis found significant differences among treatments in species composition across all localities and times, for all fish groups/families of each experiment except pomacentrids in the hard coral disturbance at Orpheus Island (Table 6.4). The ordination plots from CDA were used to identify the nature of significant differences among treatments (Fig's 6.11-6.15)

#### 6.4.3.1 Disturbance to hard corals

At Maiton Island, the likelihood ratio indicated that the first four canonical discriminants (responsible for 81% of total variance) were significantly different for labrids, while for the others, these were not significant. There was clear influence of localities due to different labrid assemblage composition. The first two canonical discriminants (53.3%) were used to display the effect of treatments at different times for NE-Reef (Fig. 6.11A). It was clear that the first canonical discriminant was responsible for the separation between living coral control (LC) and living coral disturbance (LD) which can be detected at time 2 (two weeks after manipulation) to time 5. This shifting in position of the disturbance treatment appeared to follow that of dead coral control (DC). Species responsible for the separation were *Halichoeres vrolikii* and *Thalassoma lunare* which were more common in the hard coral control and *Coris pictoides*, which was more prevalent in experimentally disturbed plots and dead coral controls (Appendix 6.3A). At SE-Reef, canonical discriminants 1 and 4 (46%)

were plotted, revealing a treatment effect beginning at time 4 and most prominent at time 5 (Fig. 6.11B). Canonical discriminant 4 was responsible for the separation, which appears to be due to *Thalassoma lunare* and *Halichoeres timorensis,* which are more abundant in living coral controls than on disturbed patches (Appendix 6.3B). The abundance of *Coris pictoides* at time 5 on disturbed plots (LD5) was also higher than for the hard coral control (LC5). The labrid assemblage on the dead coral control appeared not to change over the 5 sampling times.

For pomacentrids, likelihood ratio indicated that only the first two canonical discriminants (78.9%) are significant (later discriminants do not give different result from the previous one). However, canonical discriminant 1 and 3 (53.8%) were used to display the influence of habitat manipulation over time for NE-Reef (Fig. 6.12A). The effect of disturbance on living coral was evident in canonical discriminant 3, whereas canonical discriminant 1 distinguished the disturbance treatment from the control at time 1 only. Pomacentrids responded to the reduction in hard coral by a decreased abundance of *Dascyllus aruanus, Pomacentrus moluccensis* and *P. similis*, which after time 2 were reduced to densities similar to the dead coral control (Appendix 6.3A). There was no evidence of recovery of these pomacentrids as the study progressed. There was considerable natural temporal variation in species composition on the hard coral controls, with the dominant species changing from *Neopomacentrus azysron* to *Chromis weberi.* At the SE-Reef site, the first two canonical discriminants were plotted, in which an effect of experimental reduction in hard coral cover cannot be discerned (Fig. 6.12B). The significant pattern appeared to be due to the dead coral control and temporal variation in the disturbance and hard coral control treatments.

At Orpheus Island, CDA results for labrids indicated that only the first two canonical discriminants (40.1%) were statistically significant. However, canonical discriminants 3 and 4 (26.4%) best illustrated the effects of disturbance at Pioneer Bay, and 1 and 4 (34.4%) best illustrated the effects of disturbance at Cattle Bay (Fig. 6.13). In general, the labrid assemblages at the two localities were relatively similar (Appendix 6.4). At Pioneer Bay, the minor influence of hard coral disturbance was apparent from time 2 to 10, when these treatments could be distinguished from both the living coral and dead coral controls (Fig. 6.13A). Canonical discriminant 4 was responsible for the discrimination, and was due to a higher abundance of *Hemigymnus melapterus* in the disturbance treatment (Appendix 6.4A). At Cattle Bay there was no clear effect of the disturbance. The significant differences came from temporal variation in the dead coral and living coral controls (Fig. 6.13B).

The hard coral disturbance experiment at Orpheus Island did cause any detectable change in the species composition of the Pomacentridae. However, a clear negative trend on the effect of disturbance could be seen for *Pomacentrus moluccensis*, which underwent a decline when coral cover was reduced (Appendix 6.4).

#### 6.4.3.2 Soft coral degradation

The CDA results for labrids in the experimental disturbance to soft corals indicated only the first canonical discriminant (22.0%) was significant, which suggests only a minor effect. Canonical discriminant 1 and 4 (31.8%) best illustrated the effects of disturbance for both localities (Fig. 6.14). At Pioneer Bay, both canonical discriminants were responsible for a minor difference between soft coral disturbance treatments and soft coral controls (Fig. 6.14A). Canonical discriminant 1 was responsible for the difference at times 3 and 5 while canonical discriminant 4 responsible for the difference at time 10. Fig. 6.14A also displayed the influence of temporal variation, with the labrid assemblage in disturbance treatments exhibiting greater variation than the soft coral control and dead coral control treatments. The influence of disturbance to soft coral appeared to be an increase in the abundance of *Stethojulis strigiventer* and *Halichoeres melanurus* (Fig. 6.14A). At Cattle Bay, the discrimination between disturbance treatments and controls over time was not clear (Fig. 6.14B). In general, the influence of the reduction in soft coral on labrids at both localities was quite small (Appendix 6.5).

For pomacentrids, the first two canonical discriminants (42.9%) made a statistically significant contribution to the group separation. However, the effects of disturbance at Pioneer Bay were most apparent in canonical discriminants 1 and 4 (33.6%), canonical discriminants 3 and 4 (21.8%) showed the effects for the Cattle Bay location (Fig. 6.15). At Pioneer Bay. the influence of disturbance to soft corals was evident in canonical discriminant 1 (Fig. 6.15A). Pomacentrids responded to soft coral removal by an increase in the abundance of *Pomacentrus wardi* (Appendix 6.5A). There was also an influence of disturbance to soft corals on temporal variation, particularly at times 3, 4 and 5. After time 5, the pomacentrid assemblage in disturbance treatments converged on that of the dead coral control. The pomacentrid assemblage associated with dead coral control treatments were also different from those on the soft coral control plots (also responsible by canonical discriminant 1). At Cattle Bay, the influence of soft coral degradation could also be detected, but there was also the influence from temporal variation. The greatest difference between disturbance treatments and controls occurred between times 3 and 8, after which some recovery had occurred (Fig. 6.15B). Canonical discriminant 4 was responsible for this pattern. After time 1, the soft coral controls supported greater densities of *Neoglyphidodon melas, Chrysiptera rollandi, Pomacentrus moluccensis* and *Amblyglyphidodon curacao* than the disturbance plots (Appendix 6.5B). The pomacentrid assemblages on dead coral were clearly distinct from both soft coral controls and disturbance treatments, indicating that the disturbance treatments are not analogous to the pre-existing dead coral areas.

## 6.5 DISCUSSION

The first important conclusion from this study is that the influence of disturbances that reduce the availability of habitat-forming organisms vary at both regional and local scales, and vary depending on the fish taxa in question. Where responses occurred they were consistent with the hypothesis that these habitat resources control the abundance of many, but not all species, at least on the scale of the experiment. When limited resources are decreased, then the abundance of species dependent upon them should decline, for instance pomacentrids and live coral from this study and Sano *et al.* (1984). When other resources increase (also as a response to disturbance) the abundance of dependent species may increase, such as labrid respond to soft coral, unless there are other factors which simultaneously limit abundance (Wellington & Victor, 1985). In general, disturbance to hard coral showed a reduction in species diversity and species richness, as species dependent upon live coral were eliminated or reduced. Some increases in non-coral dwelling species occurred, but these were only detected using multivariate analyses.

### 6.5.1 HARD CORAL DEGRADATION

At Maiton Island, a significant effect of disturbance to hard corals on community parameters of all three fish groups were detected only at NE-Reef, while there was only a trend for labrids at SE-Reef. By looking at community structure, the

CDA displayed trends similar to patterns in diversity and overall abundance, indicating moderate effects of disturbance on labrids and pomacentrids at NE-Reef, while only minor effects on labrids at SE-Reef.

At Orpheus Island, a significant effect of hard coral degradation was detected on species richness and total abundance of pomacentrids at both localities. At Pioneer Bay there was a negative trend on species diversity of most fish groups. The results from CDA suggested that the effect on labrid species composition was relatively small and the effects at Cattle Bay were short term. At Orpheus Island, there was no significant effect on pomacentrids, and effects on other families were small, relative to a high background of temporal variation.

The difference between the two regions might be explained by the initial differences in reef condition. The reef at Maiton Island was considerably degraded, while that Orpheus Island was typical of undisturbed inshore reefs on the GBR. The experiment at Maiton Island was carried out in fragments of less disturbed habitat, in a landscape of coral debris and sand (ca. 80% cover). Hence, on the scale of the experiment, there was less resilience to disturbance as a result of immigration from surrounding habitat. At Orpheus island, there was greater potential for immigration and swamping of experimental effects, which may have been a factor in the greater recovery observed here.

Another factor in the specific differences in response to disturbance may be the biogeographic difference in the species composition. Although local species diversity and species richness appeared to be similar, the total abundance of all species combined at Maiton Island (mean ca. 30-40 individual/25 $m^2$) was less than at Orpheus Island (mean ca. 80-100 individual/25$m^2$). The familial structure of the fish assemblages in the two regions differed, with both labrids (44.6%) and pomacentrids (53.5%) co-dominant at Maiton Island, but pomacentrids (79.1%) dominating the fauna at Orpheus Island. Patterns of abundance of these faunas differed, with densities considerably more variable at Orpheus Island compared with Maiton Island.

Despite the different species composition, there were some similarities in the responses. The immediate effects of the coral disturbance treatment appeared to be similar in both regions, with the temporary incursion of large invertebrate feeders e.g.

*Halichoeres hortulanus* and *Coris africana* at Maiton Island, *Choerodon anchorago, C. schoenleinii, Cheilinus fasciatus,* and *Epibulus insidiator* at Orpheus Island. Juvenile recruitment into disturbed plots was dominated by sand and gravel-dwelling species especially the wrasses *Coris pictoides* and *Halichoeres argus* at Maiton Island, and the small pomacentrids *Pomacentrus adelus* and *P. wardi* at Orpheus Island.

The differences between localities within could also be explained by local differences in species-composition and habitat structure. The CDA analyses for both regions indicated that the fish assemblages always differed between the two localities. As different species/genus/families of fish respond to habitat changes differently (Sano *et al.*, 1984; 1987; Williams, 1986), this might be sufficient to explain the different local effects. Habitat structure may also be important. At Maiton Island, the experiment NE-Reef was laid out on a fragmented habitat, where much of the bottom was sand and gravel. This may have contributed to the larger effect observed here. At SE-Reef, the habitat was primarily hard reef substrata, and there was a greater potential for fish to find refuges outside disturbed plots. The basic reef habitat structure at Pioneer and Cattle Bays were more similar.

The different variables considered and methods of analysis way effect our ability to detect responses of fish to habitat-disturbance, and so our conclusions about the importance of this process. The results in this study suggested that community parameters (species richness, diversity indices) may not be particularly sensitive indicators of changes to fish communities (see also Holbrook *et al.*, 1994). This is because these parameters represent average responses of all species, with each responding in different ways. By looking at community structure (species composition and the proportional abundance of each species), any changes or shift should be detected more effectively. With the aid of appropriate multivariate statistical techniques, any complex situation by dealing with many variables can be resolved. A similar situation was found by (Dawson-Shepherd *et al.*, 1992), where fairly substantial responses of fish communities to coral mining could only be detected using multivariate procedures.

The influence of living coral degradation on fish assemblages if detected was usually negative, which might be expected from a reduction in hard coral (from approximately 40-60% to <5%). Similar declines in diversity and reductions in the abundance of coral-dwelling fishes have been recorded over this range (Bell & Galzin, 1984). Previous spatial studies on labrids have found a wide range of relationships between community parameters and habitat structure (Roberts & Ormond, 1987; Green, 1996). Results in Chapter 4 and 5 also demonstrated that the relationships between community parameters of labrids in relation to living and dead coral also varied among regions and times. Both linear (negative with living coral, but positive with dead coral) and polynomial (concave downward parabola) relationships best explained the data in different situations (Chapter 4 and 5). A decrease in living coral to 5% (with a corresponding increase in dead coral) increase dead coral to nearly 100%, resulted in an increase in some labrids that are most frequently associated with dead coral. However, changes from living to dead corals without changes in coral structure had no significant influence on labrids (Sano *et al.*, 1987).

For pomacentrids, results indicate primarily negative effects of coral degradation. There are mixed results reported for this group in the literature, with some negative effects reported (Sano *et al.*, 1987), and others not detecting any effects (Williams, 1986). Results of some spatial surveys have not detected significant quantitative relationships between pomacentrid community parameters and major habitat structure parameters (Roberts & Ormond, 1987) while a minor proportion did (Chapter 4). This may indicate the importance of field experiments for the study of pomacentrid-habitat relationships especially at community level.

### 6.5.2 SOFT CORAL DEGRADATION

The effects of disturbance to soft coral on reef fish assemblages contrasted with that for hard coral, with many positive effects recorded. At Orpheus Island, the influence of soft coral degradation was detected on all three community parameters, with increases in species richness, diversity and abundance. The effect of soft coral degradation was detected on overall fish community structure at both localities and also suggested short term effect on pomacentrids at both localities. A positive response was detected significantly for labrids at Pioneer Bay. The effect of soft coral degradation on other families was not clear.

The reason for the soft coral response is unclear. This result would not have been predicted for labrids in which positive relationships with soft coral have been observed (Chapter 4 and 5, see also Roberts & Ormond, 1987). The nature of substrata at Pioneer Bay may responsible for this result, where soft coral (*Nepthya* spp. and

*Sinularia* spp.) occupy most of the area by overgrowing living coral (mostly encrusting *Montipora* spp.). Removal of soft coral resulted in an increase in both dead coral and living coral cover as well, providing a greater range of refuges and habitat types for occupation. Although the effect was smaller at Cattle Bay, overall species diversity increased. Near 100% soft coral cover may represent less structural complexity as the habitat becomes a more uniform topography. Remove of soft coral may increase structural complexity of the habitat, which may promote a wider range of specialised species (e.g. juvenile of *Neoglyphidodon melas, Halichoeres melanurus, Siganus* spp., and *Scarus* spp.). When considering community structure, the CDA results suggested that the effects of soft coral degradation on both labrids and pomacentrids were species-specific.

Different functional groups of species appear to respond in different ways, to disturbance. Those using coral types primarily as a food source appear to show clear responses (Sano *et al.*, 1984; Williams, 1986) while species using habitat primarily as shelter appear to be less impacted (Sano *et al.*, 1987). Herbivorous fishes are particularly resilient to reductions in live coral (Sano *et al.*, 1987). Therefore, patterns may be interpreted by knowing the functional relationships between fish and the disturbed components of the habitat.

With better understanding of fish-habitat relationships, the effect of loss a particular habitat can be predicted and knowledge can be applied as a basis for ecological restoration of coral reef ecosystems. For example, effect of local degradation of living coral on fish can occur as a continuous directional change through time (phase shift). In terms of positive effects, removing of some habitat attributes such as soft coral or turf algae to increase habitat complexity and the availability of hard substrata, may increase the chance of recovery of hard corals and associated fishes.

### 6.5.3 CONCLUSION

The disturbance experiments indicated that biotic habitat structure had an influence on the structure of reef fish assemblages. The relationships observed depended on the species, family and guild in question. Despite regional and local variation in responses, some typical patterns emerged, including negative responses to hard coral disturbance and positive responses to disturbance of soft corals. In general, analytical methods that described changes in species composition were more sensitive indicators of responses to disturbance. Although experiments were small in scale, when linked to large-scale descriptions of fish-habitat relationships, a scale-dependent picture of responses to disturbance may emerge.

Table. 6.1. ANOVA and SNK test results of three community parameters (species diversity, species richness, total abundance) showing the response of coral reef fish assemblages to habitat changes in the living coral degradation experiment conducted at two localities around Maiton Island, Phuket, Thailand. Before manipulation (Before) used a one way ANOVA while after manipulation (After) used a two way ANOVA of time and treatment effect (all fixed factors). LC = Living coral control, LD = Living coral degradation, and DC = Dead coral control. Order of treatments (where presented) are from maximum to minimum value. * = significant at $P < 0.05$, ns = non-significant, underline bar indicate non-significant different (SNK) between treatments.

| Fish group | Community parameter | NE-Reef Before | After | SE-Reef Before | After |
|---|---|---|---|---|---|
| Overall | Diversity | treatment$^{ns}$ | treatment*<br>LC DC LD | treatment$^{ns}$ | treatment*<br>LC LD DC |
| | Richness | treatment$^{ns}$ | treatment*<br>LC DC LD | treatment$^{ns}$ | treatment*<br>LD LC DC |
| | Abundance | treatment*<br>LC LD DC | treatment*<br>LC DC LD | treatment$^{ns}$ | time*, treatment*<br>LC LD DC |
| Labridae | Diversity | treatment$^{ns}$ | All factors $^{ns}$ | treatment$^{ns}$ | treatment*<br>LD LC DC |
| | Richness | treatment$^{ns}$ | treatment*<br>LC DC LD | treatment$^{ns}$ | treatment*<br>LD LC DC |
| | Abundance | treatment*<br>LC LD DC | treatment*<br>LC DC LD | treatment$^{ns}$ | All factors $^{ns}$ |
| Pomacentridae | Diversity | treatment$^{ns}$ | treatment*<br>LC DC LD | treatment$^{ns}$ | All factors $^{ns}$ |
| | Richness | treatment$^{ns}$ | treatment*<br>LC DC LD | treatment$^{ns}$ | All factors $^{ns}$ |
| | Abundance | treatment*<br>LC LD DC | treatment*<br>LC DC LD | treatment*<br>LC LD DC | time*, treatment*<br>LC LD DC |

Table. 6.2. ANOVA and SNK test results of three community parameters (species diversity, species richness, total abundance) showing the response of coral reef fish assemblages to habitat changes in the living coral degradation experiment conducted at two localities around Orpheus Island, Great Barrier Reef, Australia. Before manipulation (Before) used a one way ANOVA while after manipulation (After) used a two way ANOVA of time and treatment effect (all fixed factors). LC = Living coral control, LD = Living coral degradation, and DC = Dead coral control. Order of treatments (where presented) are from maximum to minimum value. * = significant at $P < 0.05$, ns = non-significant, underline bar. indicate non-significant different (SNK) between treatments.

| Fish group | Community parameter | Pioneer Bay<br>Before | After | Cattle Bay<br>Before | After |
|---|---|---|---|---|---|
| Overall | Diversity | treatment$^{ns}$ | time*, treatment*<br>LC LD DC | treatment$^{ns}$ | time*, treatment*<br>LC LD DC |
| | Richness | treatment$^{ns}$ | time*, treatment*<br>LC LD DC | treatment*<br>LC LD DC | time*, treatment*<br>LC LD DC |
| | Abundance | treatment*<br>LD LC DC | treatment*<br>LC LD DC | treatment$^{ns}$ | time* |
| Labridae | Diversity | treatment$^{ns}$ | time* | treatment$^{ns}$ | time* |
| | Richness | treatment$^{ns}$ | time* | treatment$^{ns}$ | time* |
| | Abundance | treatment$^{ns}$ | Time* | treatment$^{ns}$ | time* |
| Pomacentridae | Diversity | treatment$^{ns}$ | treatment*<br>LC LD DC | treatment$^{ns}$ | time*, treatment*<br>LC LD DC |
| | Richness | treatment$^{ns}$ | time*, treatment*<br>LC LD DC | treatment*<br>LC LD DC | time*, treatment*<br>LC DC LD |
| | Abundance | treatment*<br>LD LC DC | treatment*<br>LC LD DC | treatment$^{ns}$ | treatment*<br>LC DC LD |
| Other families | Diversity | treatment$^{ns}$ | time*, treatment*<br>LC LD DC | treatment*<br>LC LD DC | time*, treatment*<br>LC DC LD |
| | Richness | treatment$^{ns}$ | time*, treatment*<br>LC LD DC | treatment*<br>LC LD DC | time*, treatment*<br>LC LD DC |
| | Abundance | treatment$^{ns}$ | time*, treatment*<br>LC LD DC | treatment*<br>LC LD DC | time*, treatment*<br>LC LD DC |

Table. 6.3. ANOVA and SNK test results of three community parameters (species diversity, species richness, total abundance) showing the response of coral reef fish assemblages to habitat changes in the soft coral degradation experiment conducted at two localities around Orpheus Island, Great Barrier Reef, Australia. Before manipulation (Before) used a one way ANOVA while after manipulation (After) used a two way ANOVA of time and treatment effect (all fixed factors). SC = Soft coral control, SD = Soft coral degradation, and DC = Dead coral control. Order of treatments (where presented) are from maximum to minimum value. * = significant at $P < 0.05$, ns = non-significant. Underline bar indicate non-significant different (SNK test) between treatments.

| Fish group | Community parameter | Pioneer Bay | | Cattle Bay | |
|---|---|---|---|---|---|
| | | Before | After | Before | After |
| Overall | Diversity | treatment$^{ns}$ | time*, timeXtreatment* | treatment*<br>SD SC DC | time* |
| | Richness | treatment$^{ns}$ | time*, treatment*<br>SD DC SC | treatment$^{ns}$ | time* |
| | Abundance | treatment$^{ns}$ | time*, treatment*<br>SD SC DC | treatment$^{ns}$ | time* |
| Labridae | Diversity | treatment$^{ns}$ | time*, treatment*<br>SD DC SC | treatment$^{ns}$ | treatment$^{ns}$ |
| | Richness | treatment$^{ns}$ | time*, treatment*<br>SD DC SC | treatment$^{ns}$ | time*, treatment*<br>DC SD SC |
| | Abundance | treatment$^{ns}$ | time*, treatment*<br>DC SD SC | treatment$^{ns}$ | time*, treatment*<br>DC SD SC |
| Pomacentridae | Diversity | treatment$^{ns}$ | timeXtreatment* | treatment$^{ns}$ | time* |
| | Richness | treatment$^{ns}$ | treatment$^{ns}$ | treatment$^{ns}$ | time* |
| | Abundance | treatment$^{ns}$ | treatment*<br>SD SC DC | treatment$^{ns}$ | time* |
| Other families | Diversity | treatment$^{ns}$ | time* | treatment$^{ns}$ | time* |
| | Richness | treatment$^{ns}$ | time* | treatment$^{ns}$ | time* |
| | Abundance | treatment$^{ns}$ | time* | treatment*<br>SC SD DC | time*, treatment*<br>SD SC DC |

Table 6.4. Results of Multivariate significance tests for canonical discriminant analysis of the effect of treatment, time and locality on different fish assemblages from two experiments carried out at two regions.

| Experiment | Region | Fish assemblage | df | Pillai's trace | F | P |
|---|---|---|---|---|---|---|
| Living coral Degradation | Maiton Island | Labridae | 364, 756 | 6.49 | 1.80* | <0.001 |
| | | Pomacentridae | 182, 378 | 3.21 | 1.76* | <0.001 |
| | Orpheus Island | Labridae | 590, 1800 | 2.99 | 1.30* | <0.001 |
| | | Pomacentridae | 885, 2700 | 3.91 | 1.08ns | 0.091 |
| | | Other families | 236, 720 | 1.27 | 1.41* | <0.001 |
| Soft coral Degradation | Orpheus Island | Labridae | 531, 1620 | 2.53 | 1.19* | 0.006 |
| | | Pomacentridae | 885, 2700 | 4.30 | 1.23* | <0.001 |
| | | Other families | 236, 720 | 1.50 | 1.84* | <0.001 |

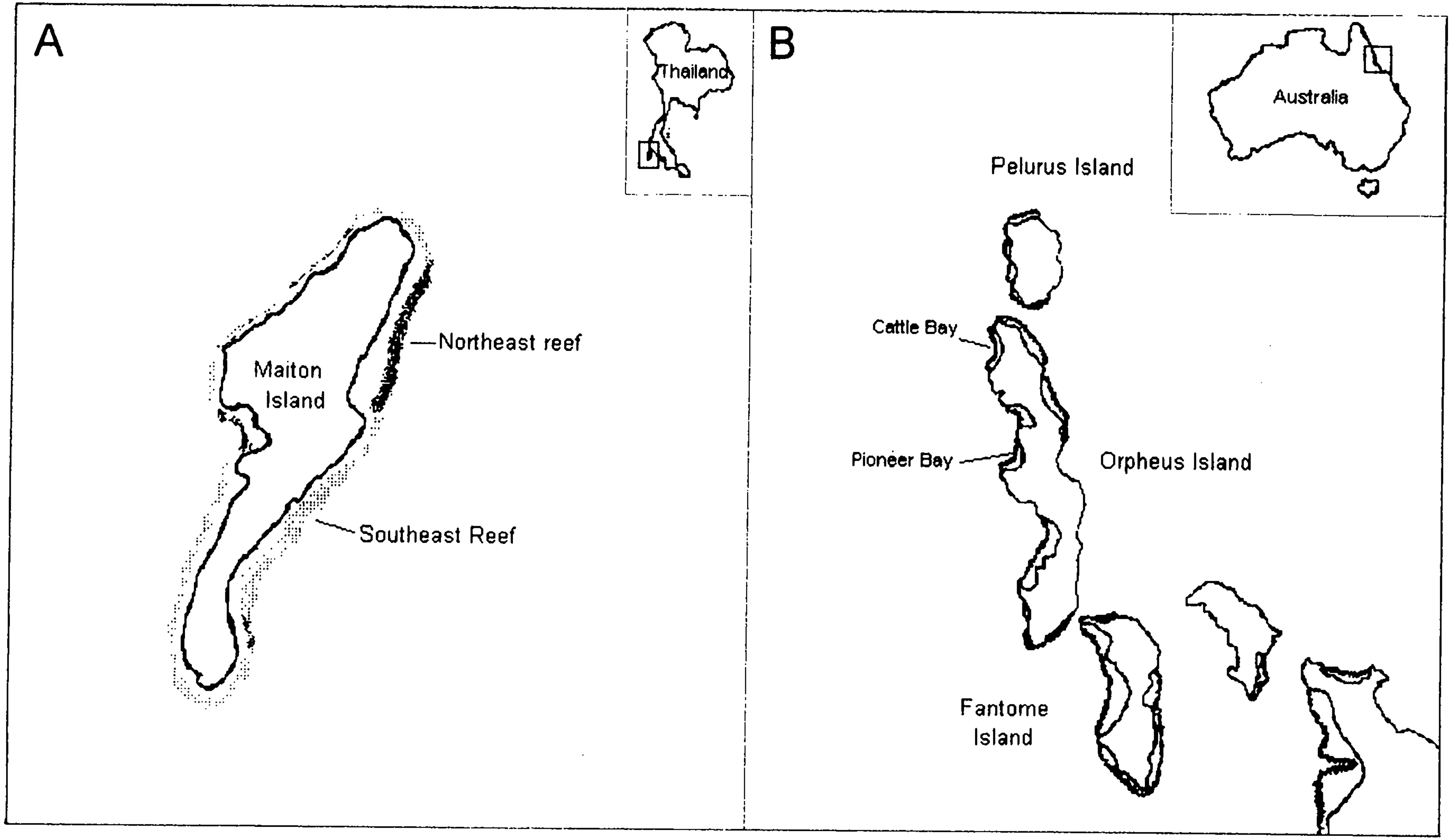


Fig. 6.1. Maps showing study areas, A) Maiton Island - Phuket, Thailand and B) Orpheus Island - Central Great Barrier Reef, Australia. In each region, two localities (Northeast reef, Southeast reef and Cattle Bay, Pioneer Bay respectively) were selected for conducting experiment.

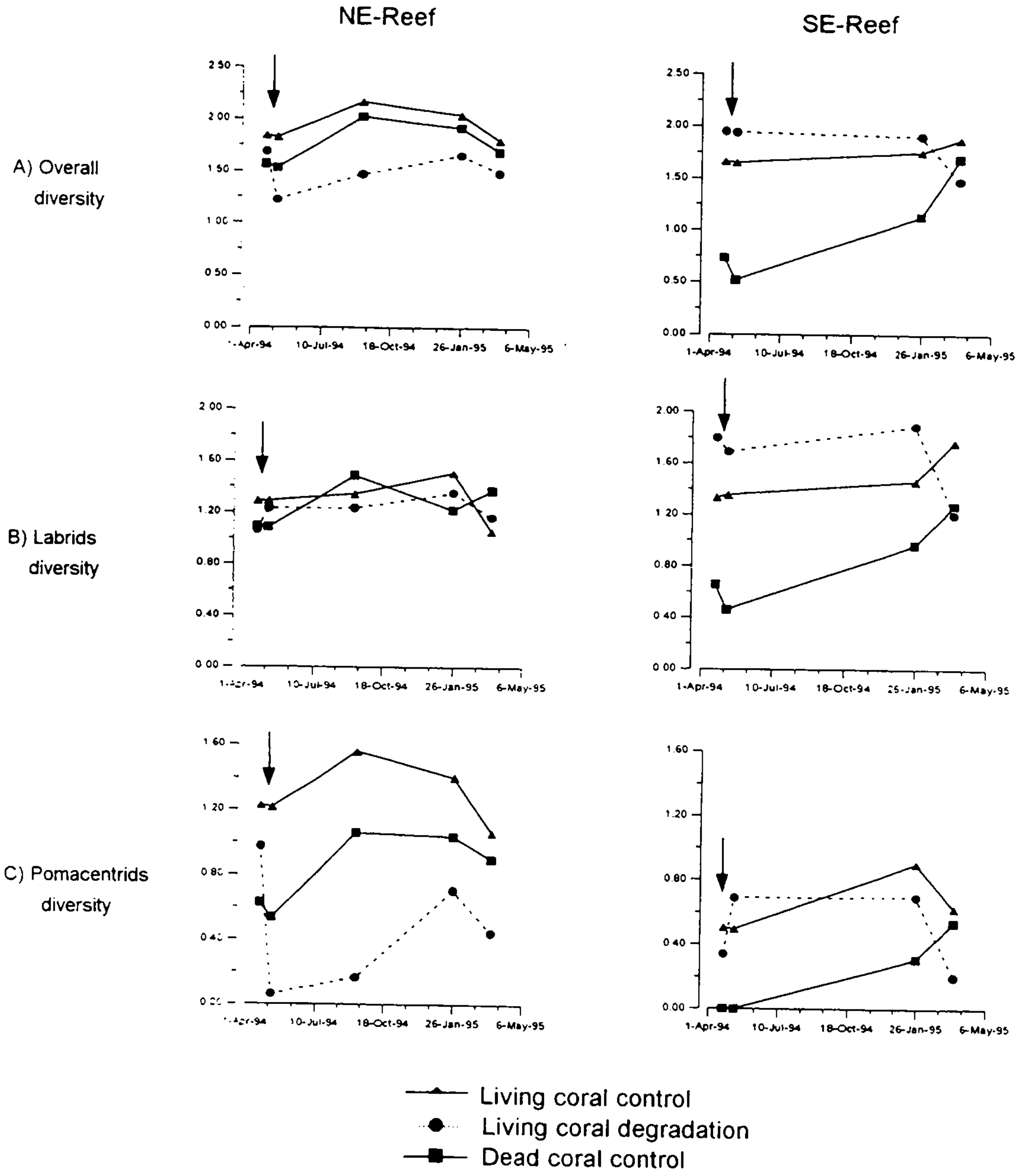


Fig. 6.2. Species diversity (mean diversity/25 sq. m) of A) Overall fish, B) Labrid, and C) Pomacentrid assemblages during living coral degradation experiments at two localities on Maiton Island. Arrows indicate time of manipulation.

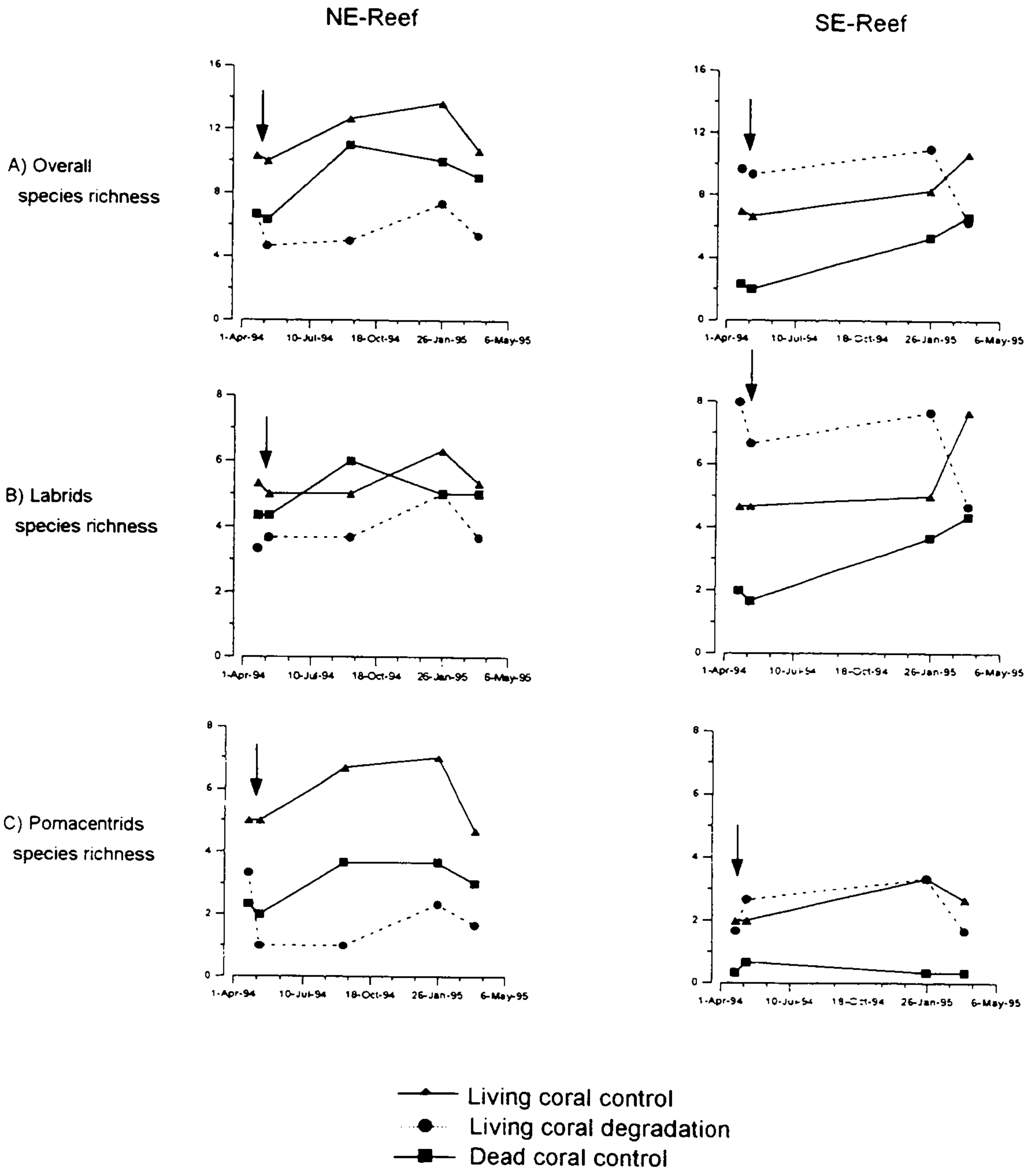


Fig. 6.3. Species richness (mean no. species/25 sq. m) of A) Overall fish. B) Labrid, and C) Pomacentrid assemblages during living coral degradation experiments at two localities on Maiton Island. Arrows indicate time of manipulation.

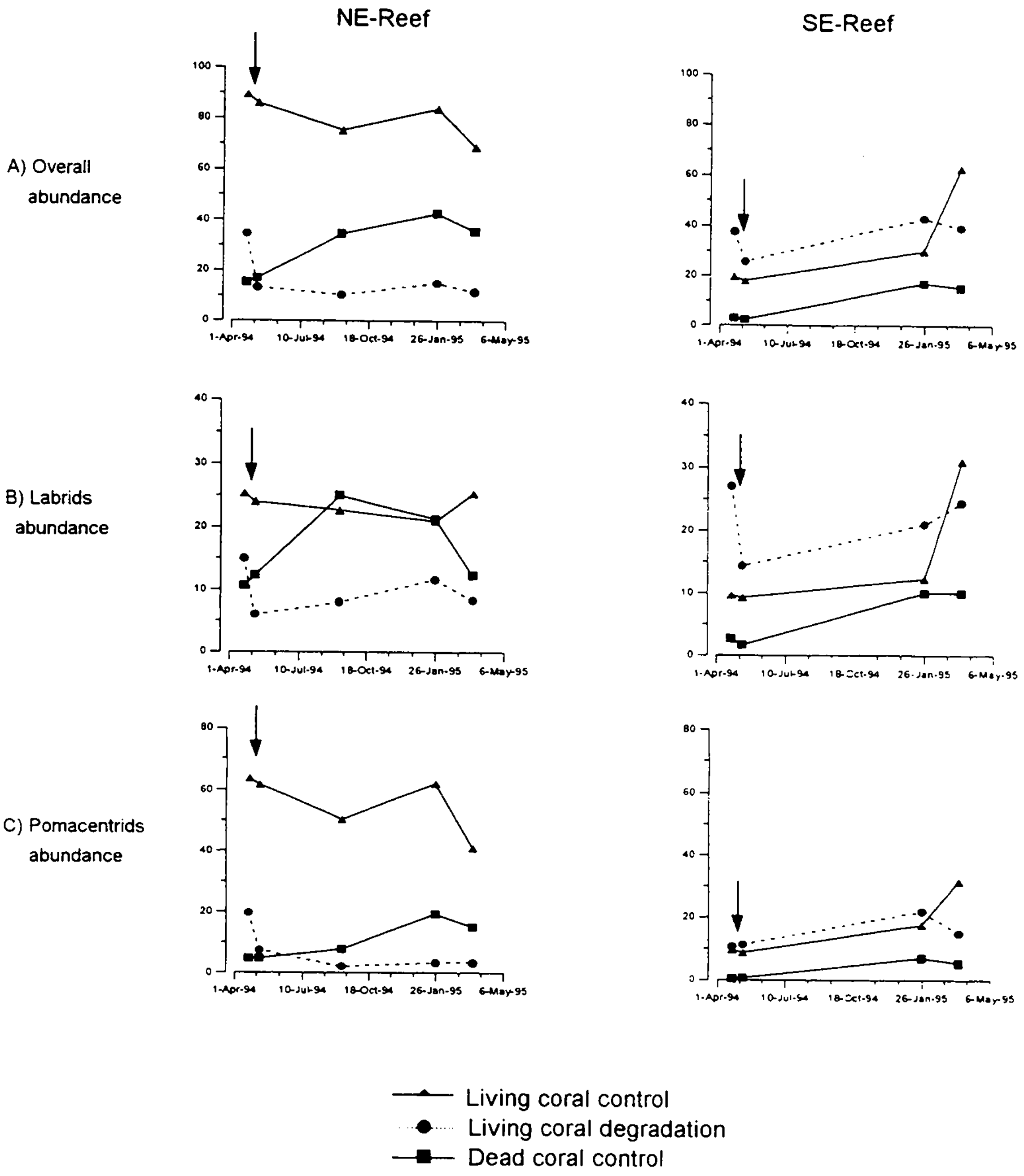


Fig. 6.4. Total abundance (mean no. fish/25 sq. m) of A) Overall fish, B) Labrid, and C) Pomacentrid assemblages during living coral degradation experiments at two localities on Maiton Island. Arrows indicate time of manipulation.

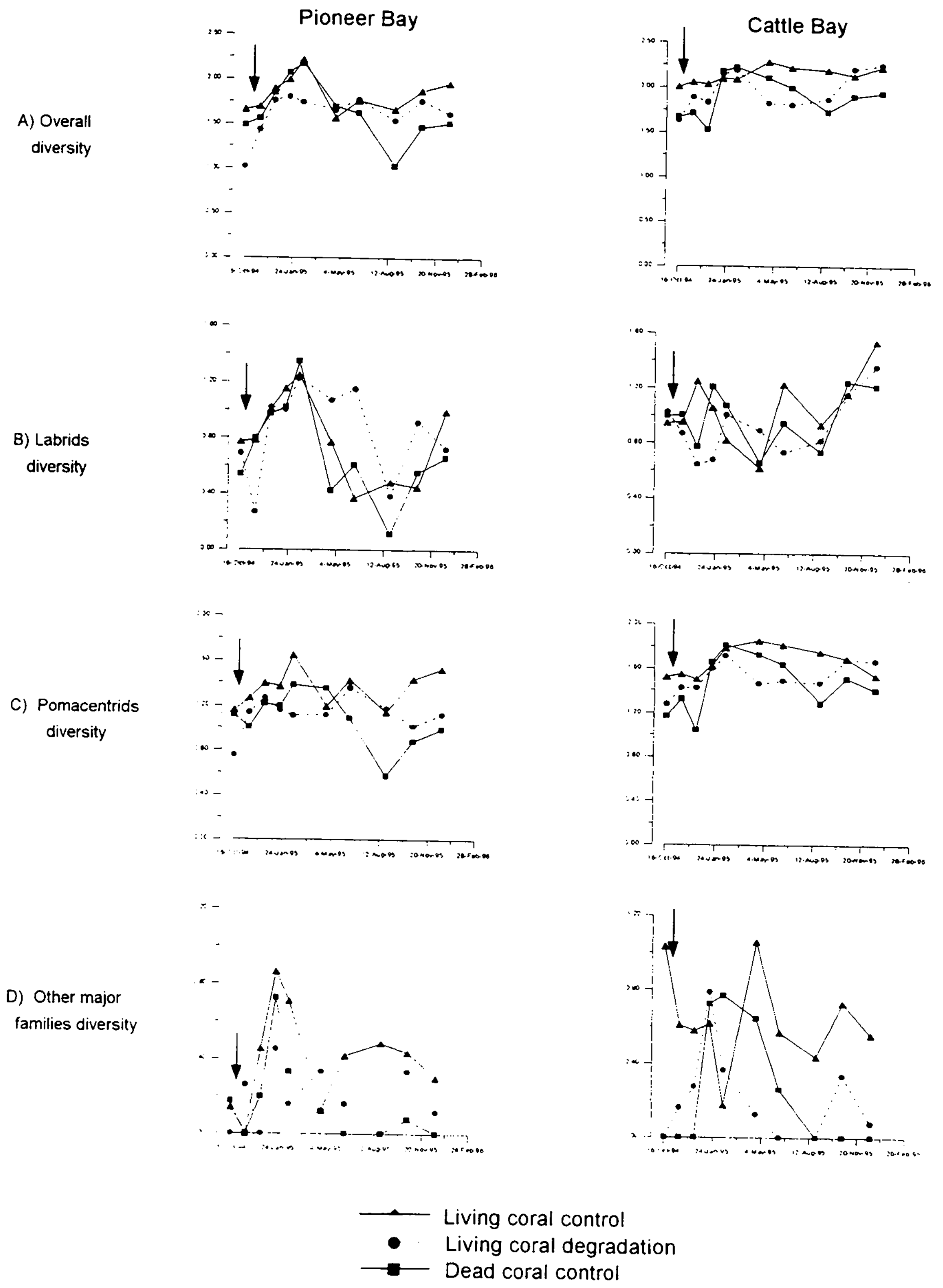


Fig. 6.5. Species diversity (mean diversity/25 sq. m) of A) Overall fish, B) Labrid, C) Pomacentrid and D) Other major families during living coral degradation experiments at two localities on Orpheus Island. Arrows indicate time of manipulation.

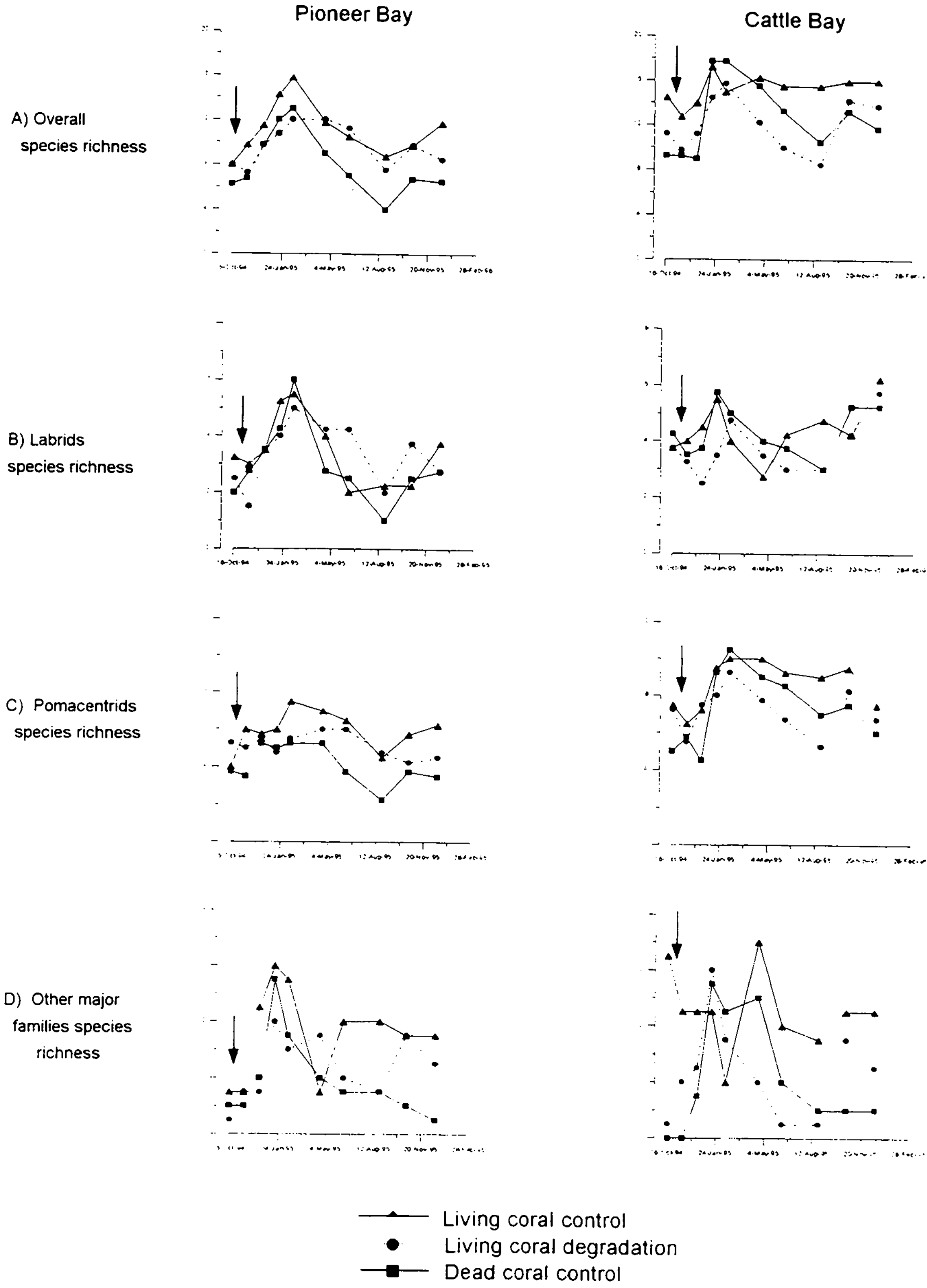


Fig. 6.6. Species richness (mean no. species/25 sq. m) of A) Overall fish, B) Labrid, C) Pomacentrid and D) Other major families during living coral degradation experiments at two localities on Orpheus Island. Arrows indicate time of manipulation.

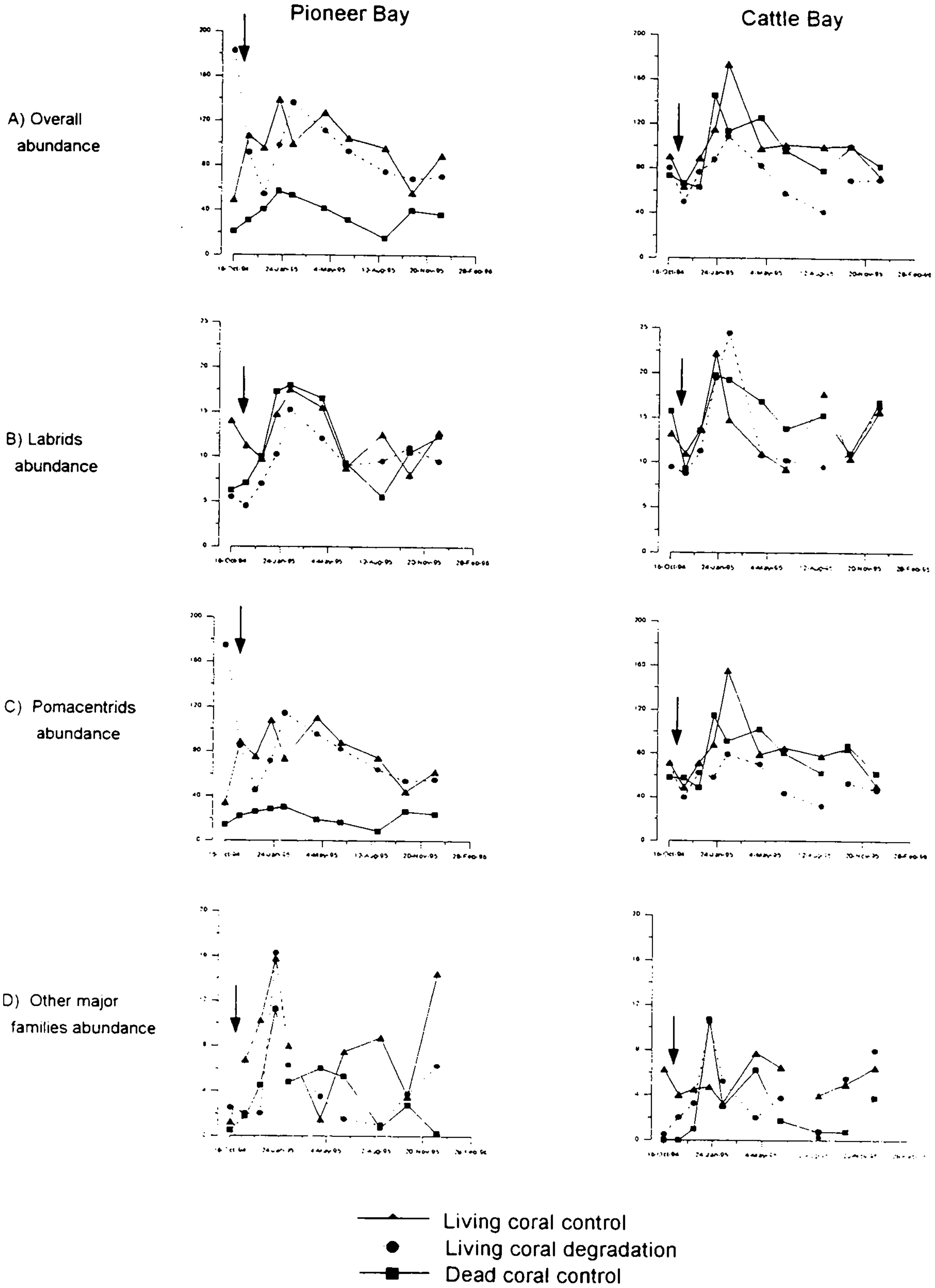


Fig. 6.7. Total abundance (mean no. fish/25 sq. m) of A) Overall fish, B) Labrid, C) Pomacentrid and D) Other major families during living coral degradation experiments at two localities on Orpheus Island. Arrows indicate time of manipulation.

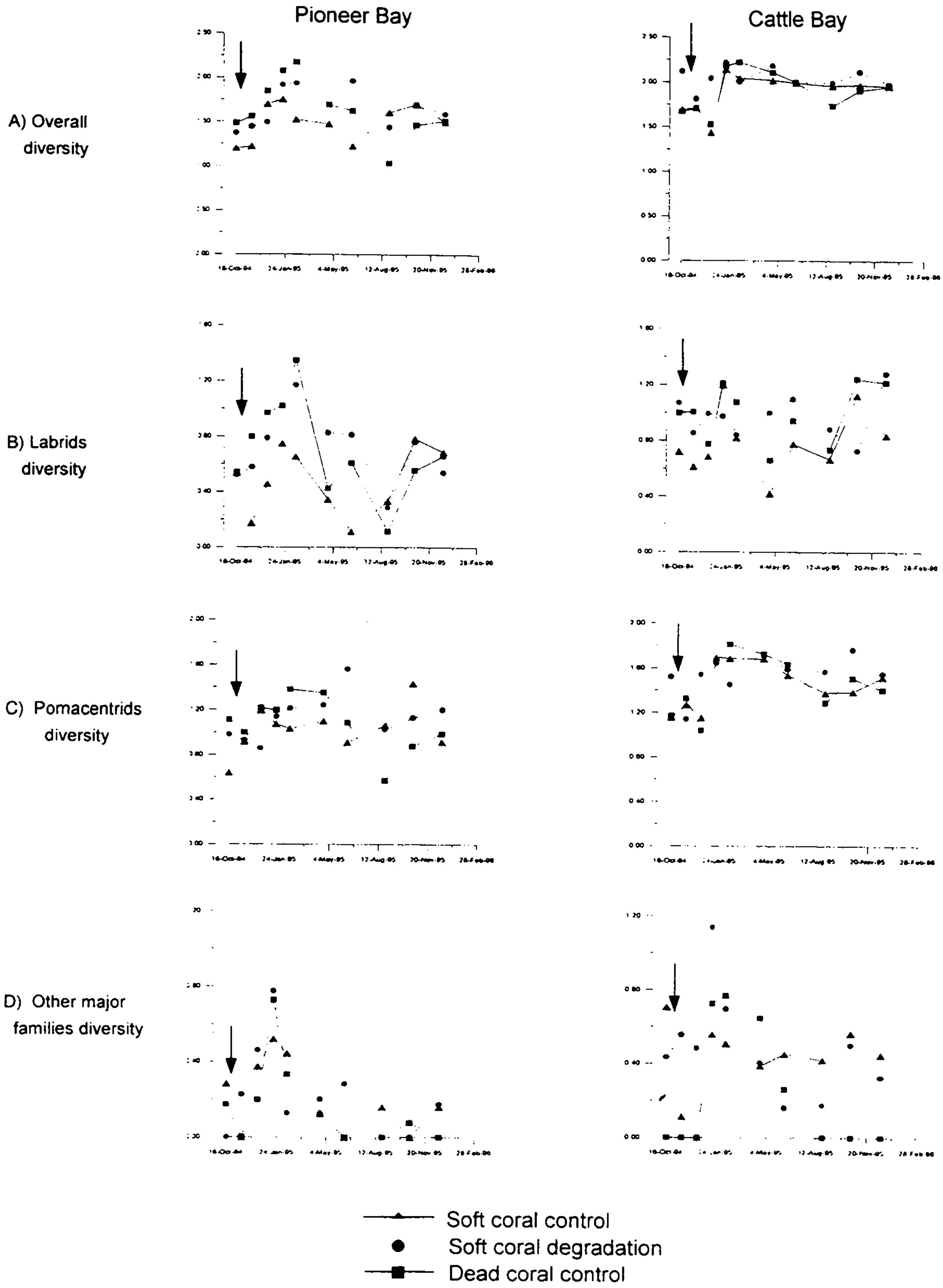


Fig. 6.8. Species diversity (mean diversity/25 sq. m) of A) Overall fish, B) Labrid, C) Pomacentrid and D) Other major families during soft coral degradation experiments at two localities on Orpheus Island. Arrows indicate time of manipulation.

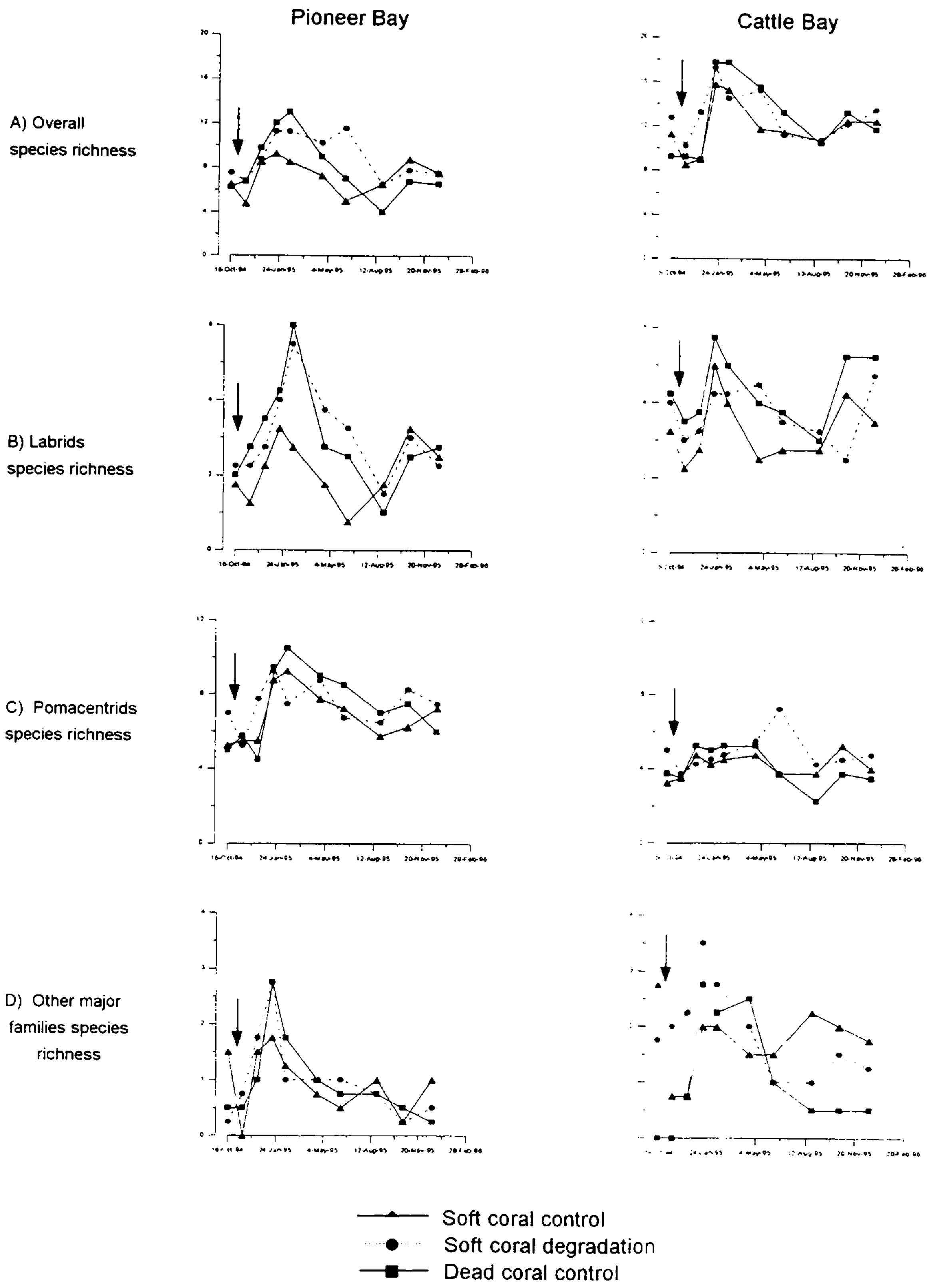


Fig. 6.9. Species richness (mean no. species/25 sq. m) of A) Overall fish, B) Labrid, C) Pomacentrid and D) Other major families during soft coral degradation experiment at two localities on Orpheus Island. Arrows indicate time of manipulation.

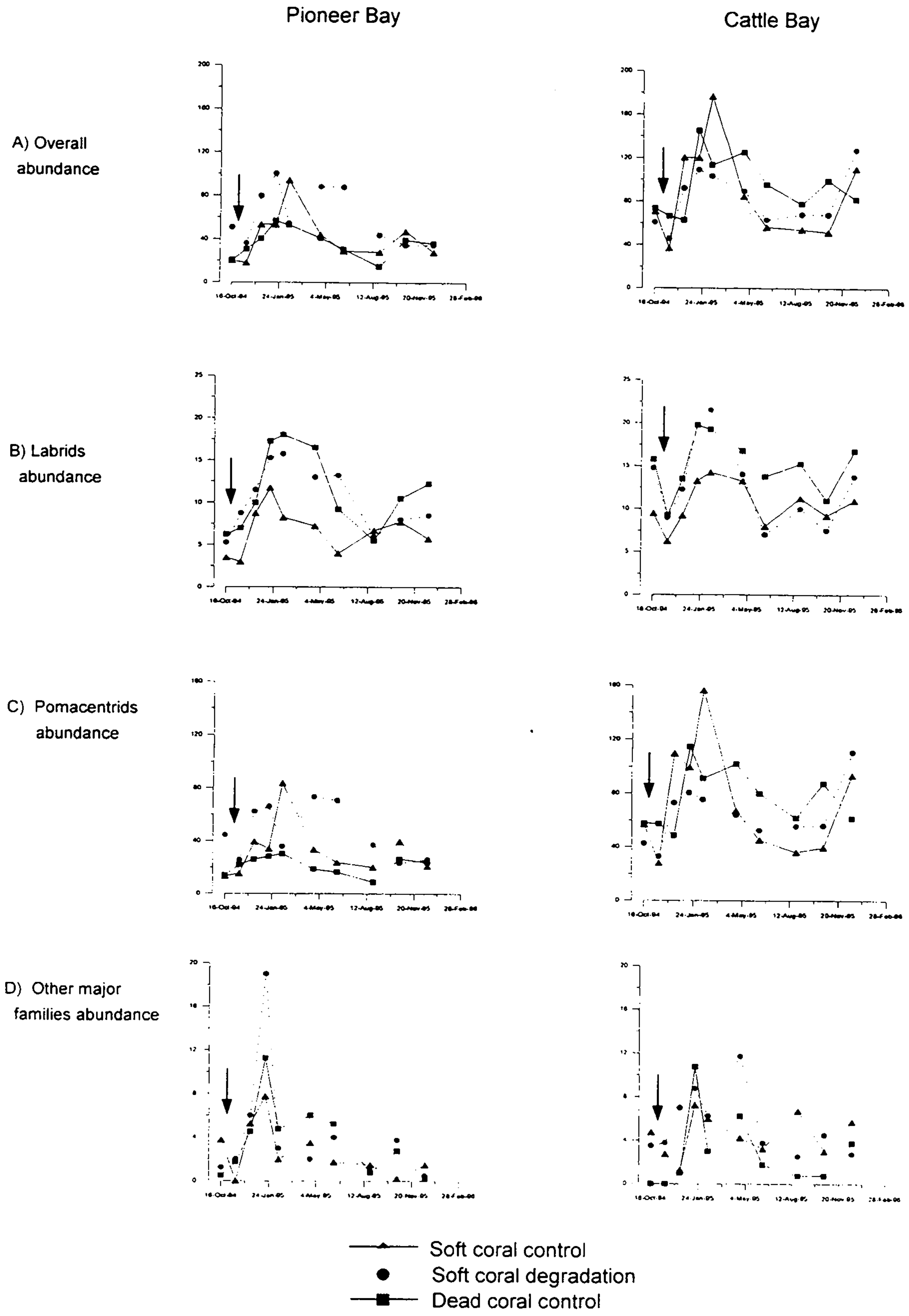


Fig. 6.10. Total abundance (mean no fish/25 sq. m) of A) Overall fish, B) Labrid, C) Pomacentrid and D) Other major families during soft coral degradation experiments at two localities on Orpheus Island. Arrows indicate time of manipulation.

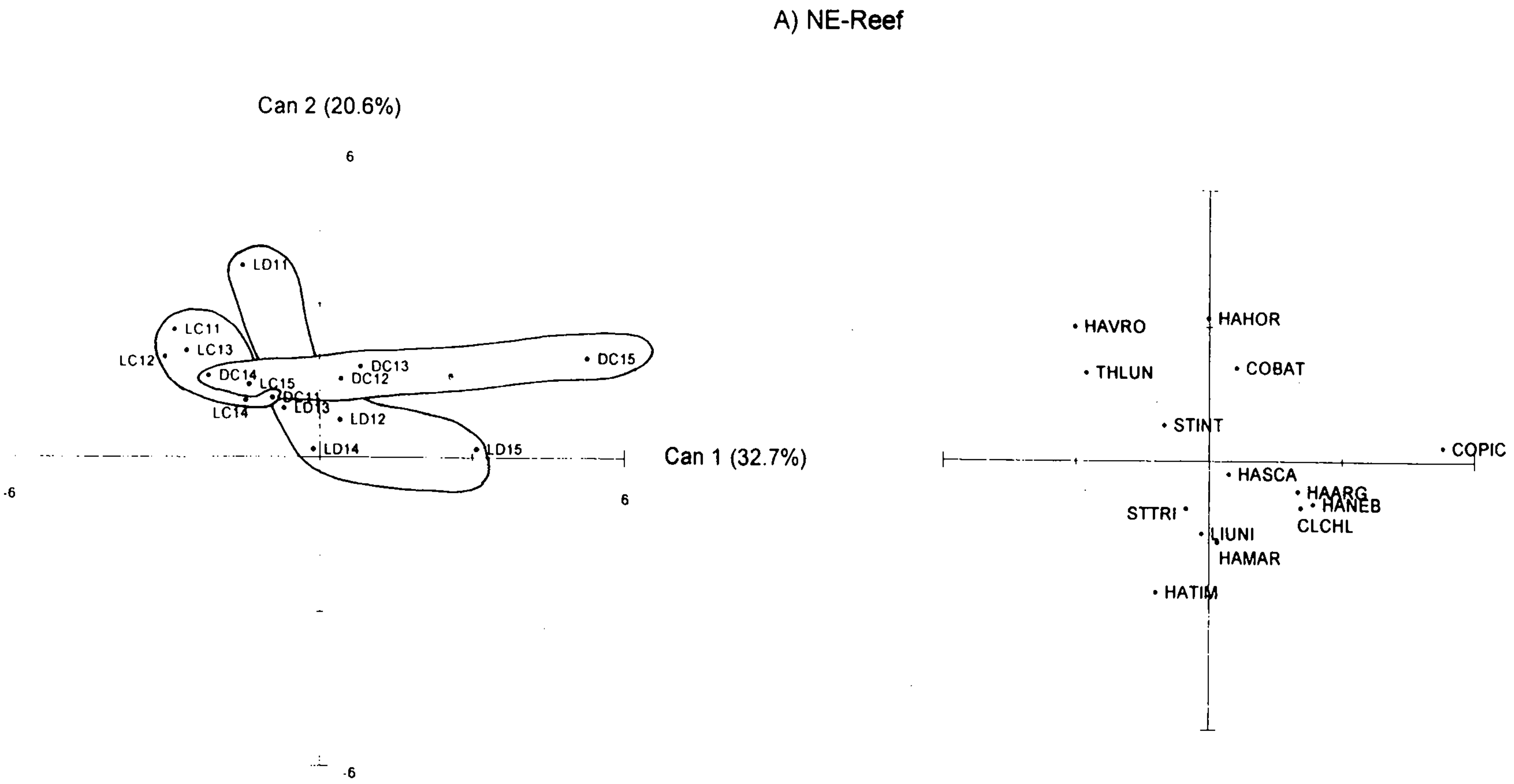


Fig. 6.11A. Ordination plots from CDA showing labrid assemblage responses to habitat manipulation in the living coral degradation experiment at NE-Reef on Maiton Island. Treatment code name indicates three factors first two letters represent treatment, LC = Living coral control, LD = Living coral degradation and DC = DC = Dead coral control; the first number represents localities, 1 = NE-Reef and 2 = SE-Reef; last number represents sampling time. Fish code name are as described in Appendix 6.1.

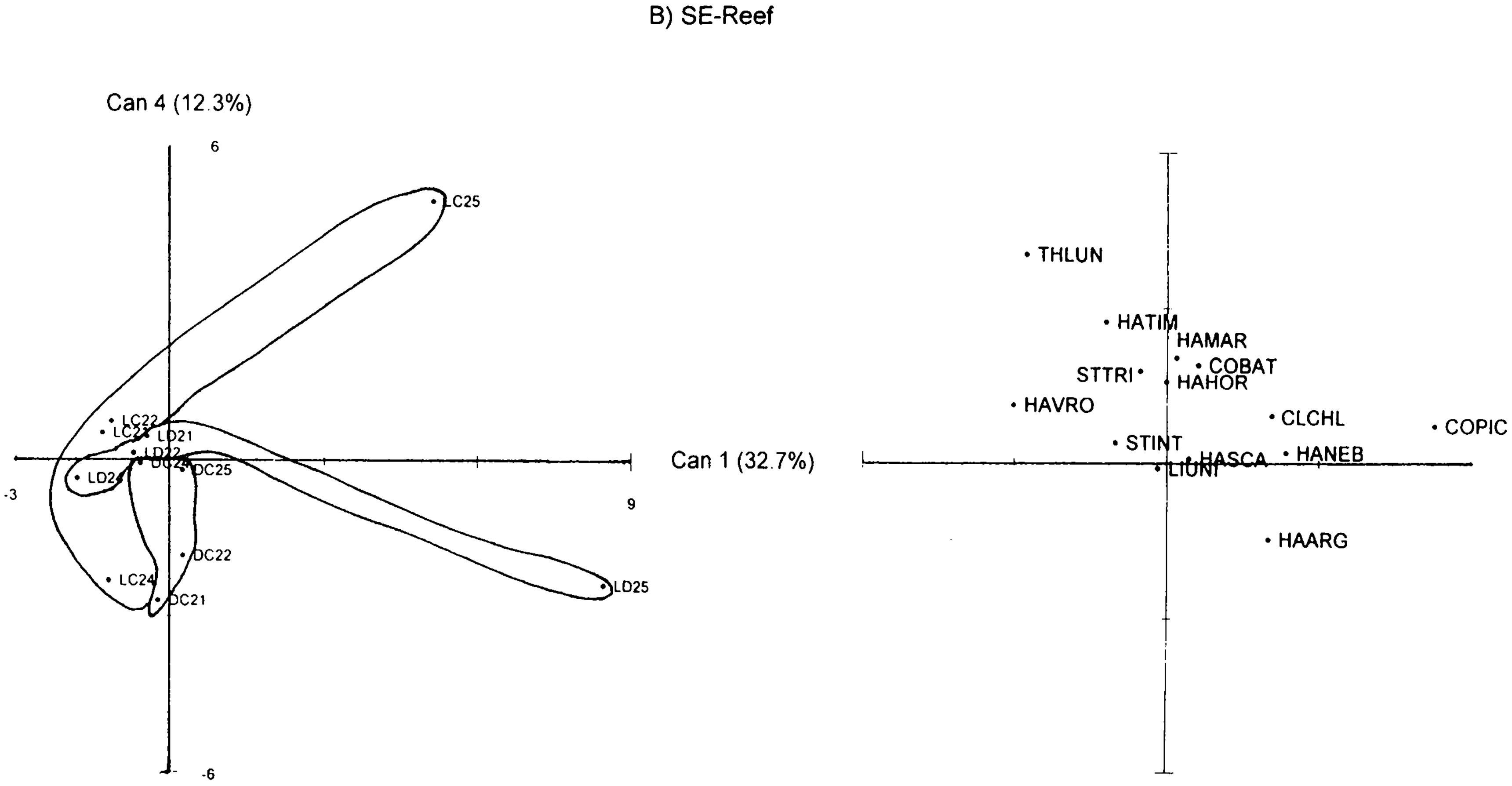


Fig. 6.11B. Ordination plots from CDA showing labrid assemblage responses to habitat manipulation in the living coral degradation experiment at SE-Reef on Maiton Island. Treatment code name indicates three factors first two letters represent treatment, LC = Living coral control, LD = Living coral degradation and DC = DC = Dead coral control; the first number represents localities, 1 = NE-Reef and 2 = SE-Reef; last number represents sampling time. Fish code name are as described in Appendix 6.1.

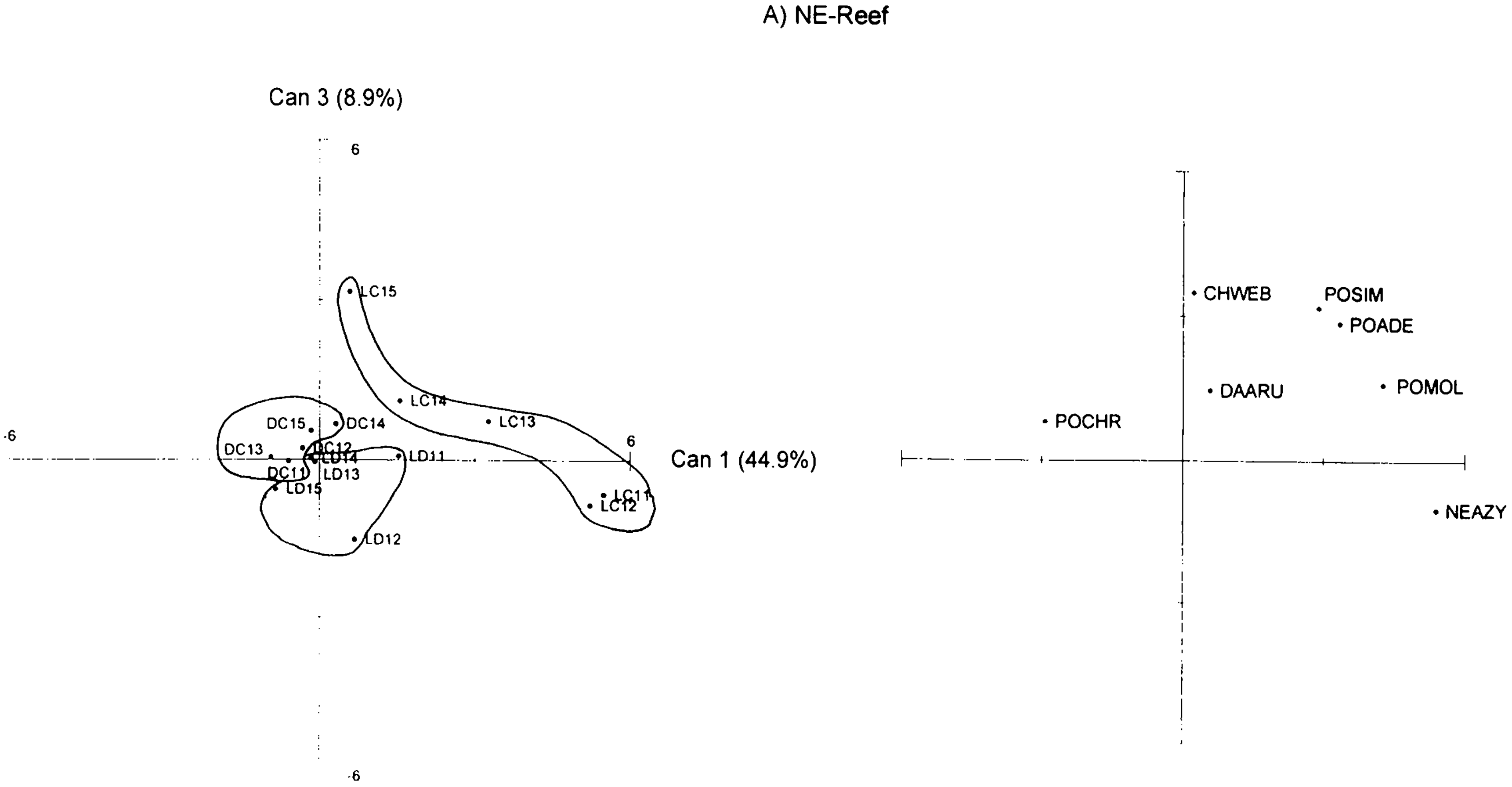


Fig. 6.12A. Ordination plots from CDA showing responses of the pomacentrid assemblage to habitat manipulation in the living coral degradation experiments at NE-Reef on Maiton Island. Code names as in Fig. 6.11.

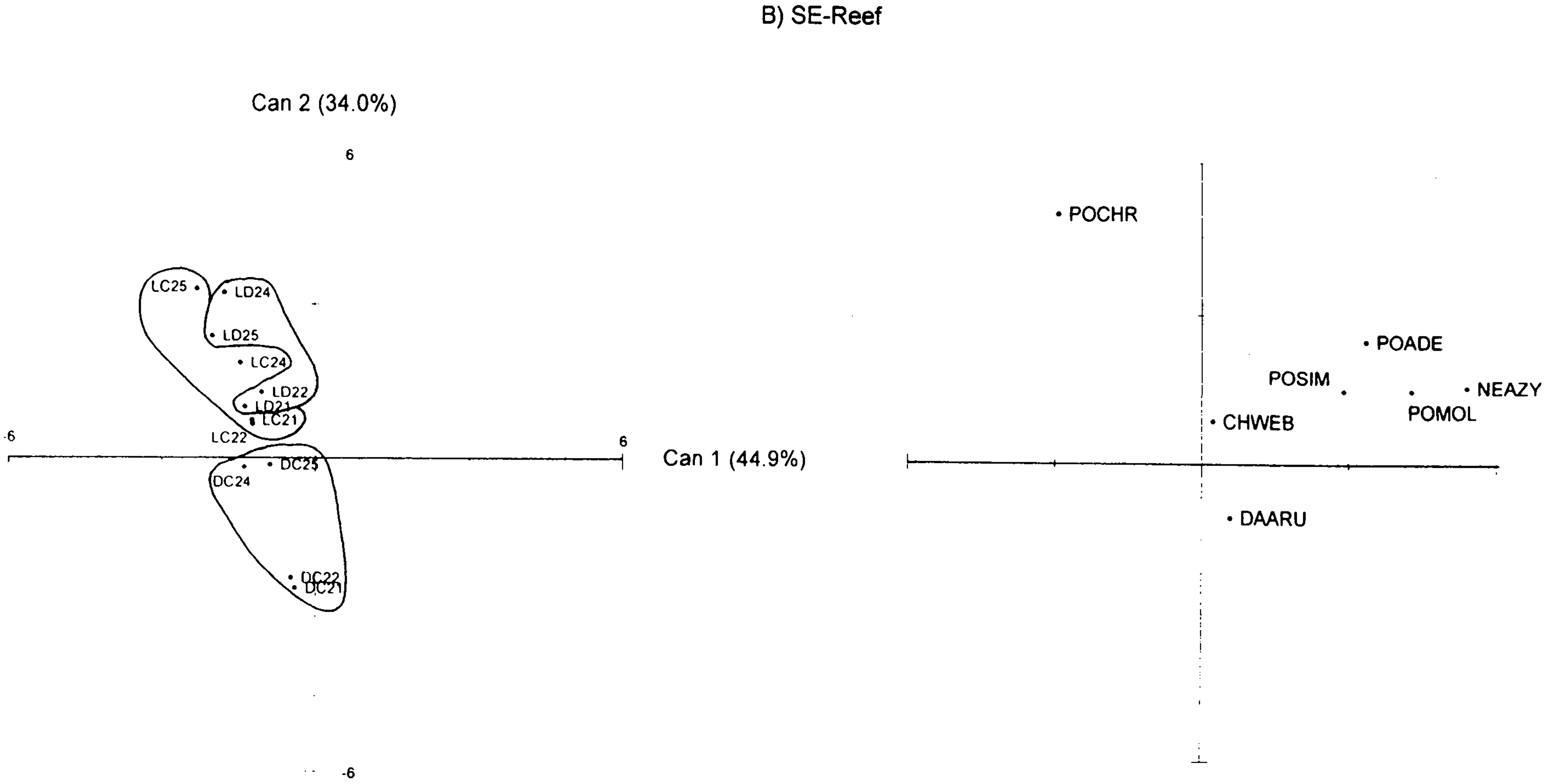


Fig. 6.12B. Ordination plots from CDA showing responses of the pomacentrid assemblage to habitat manipulation in the living coral degradation experiments at SE-Reef on Maiton Island. Code names as in Fig. 6.11.

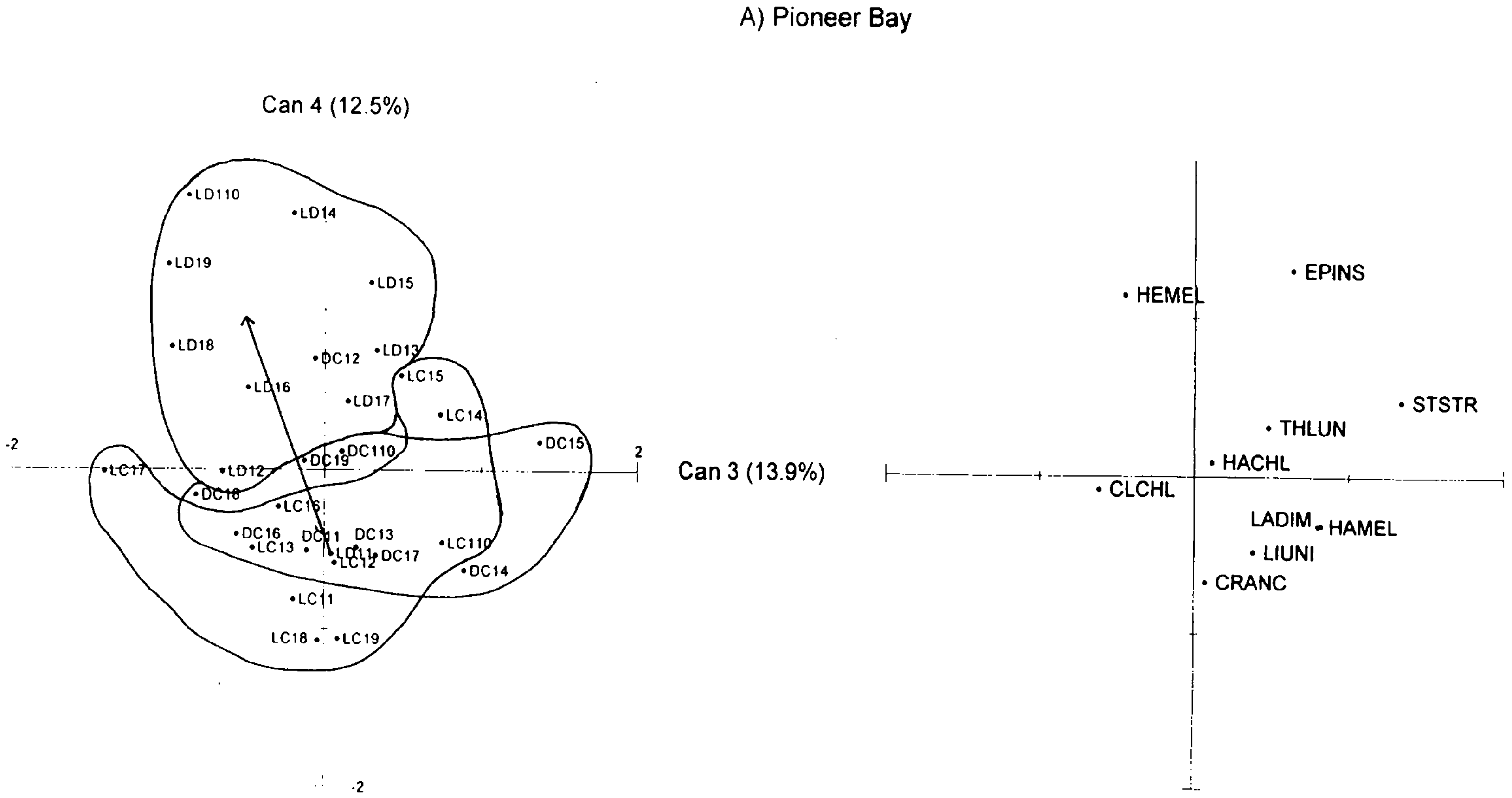


Fig. 6.13A. Ordination plots from CDA showing responses of the labrid assemblage to habitat manipulation in the living coral degradation experiments at Pioneer Bay, Orpheus Island. Treatment code names as in Fig. 6.11 except that the first number represents the localities, 1 = Pioneer Bay and 2 = Cattle Bay.

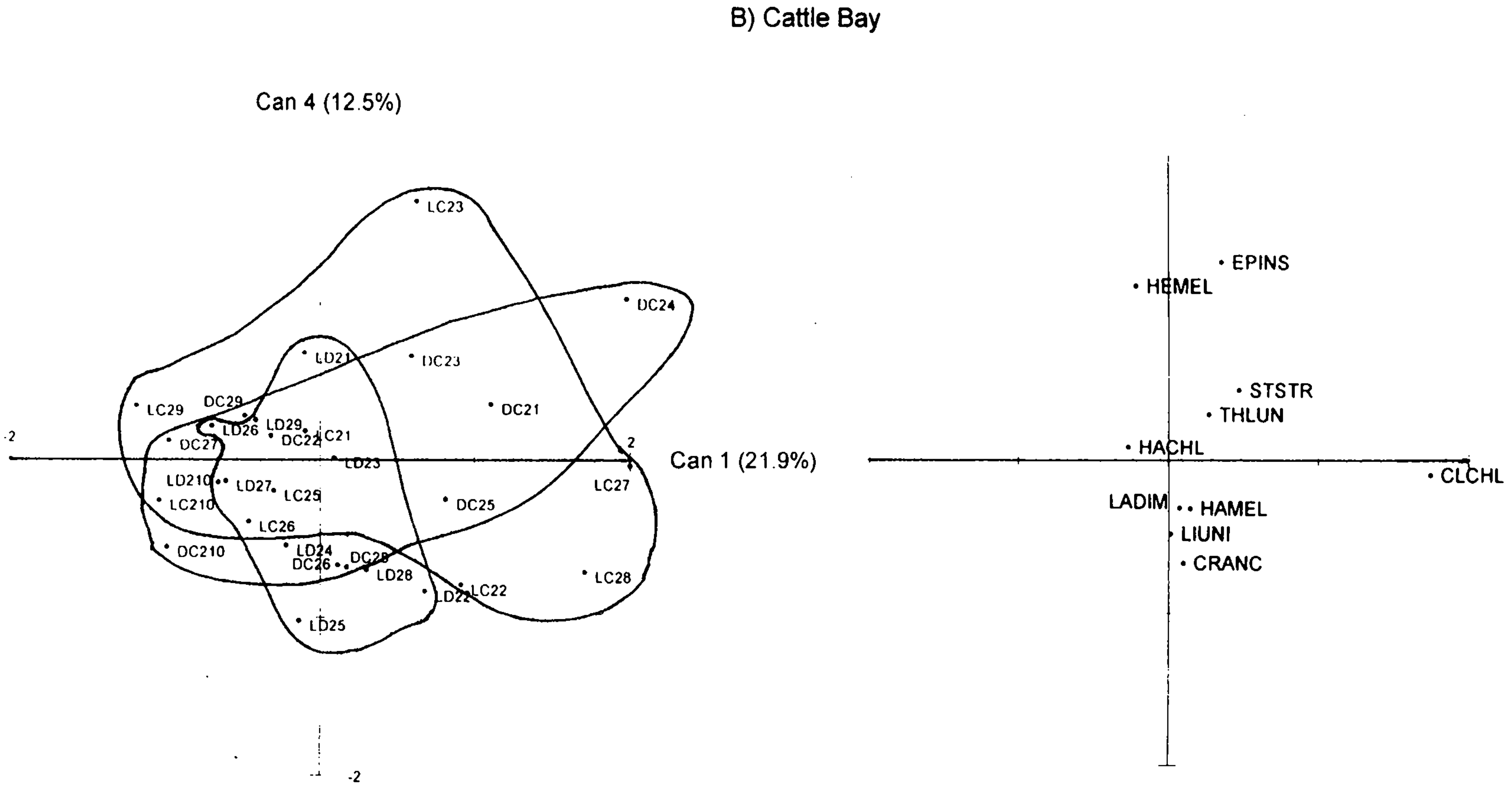

B) Cattle Bay

Fig. 6.13B. Ordination plots from CDA showing responses of the labrid assemblage to habitat manipulation in the living coral degradation experiments at Cattle Bay, Orpheus Island. Treatment code names as in Fig. 6.11 except that the first number represents the localities, 1 = Pioneer Bay and 2 = Cattle Bay.

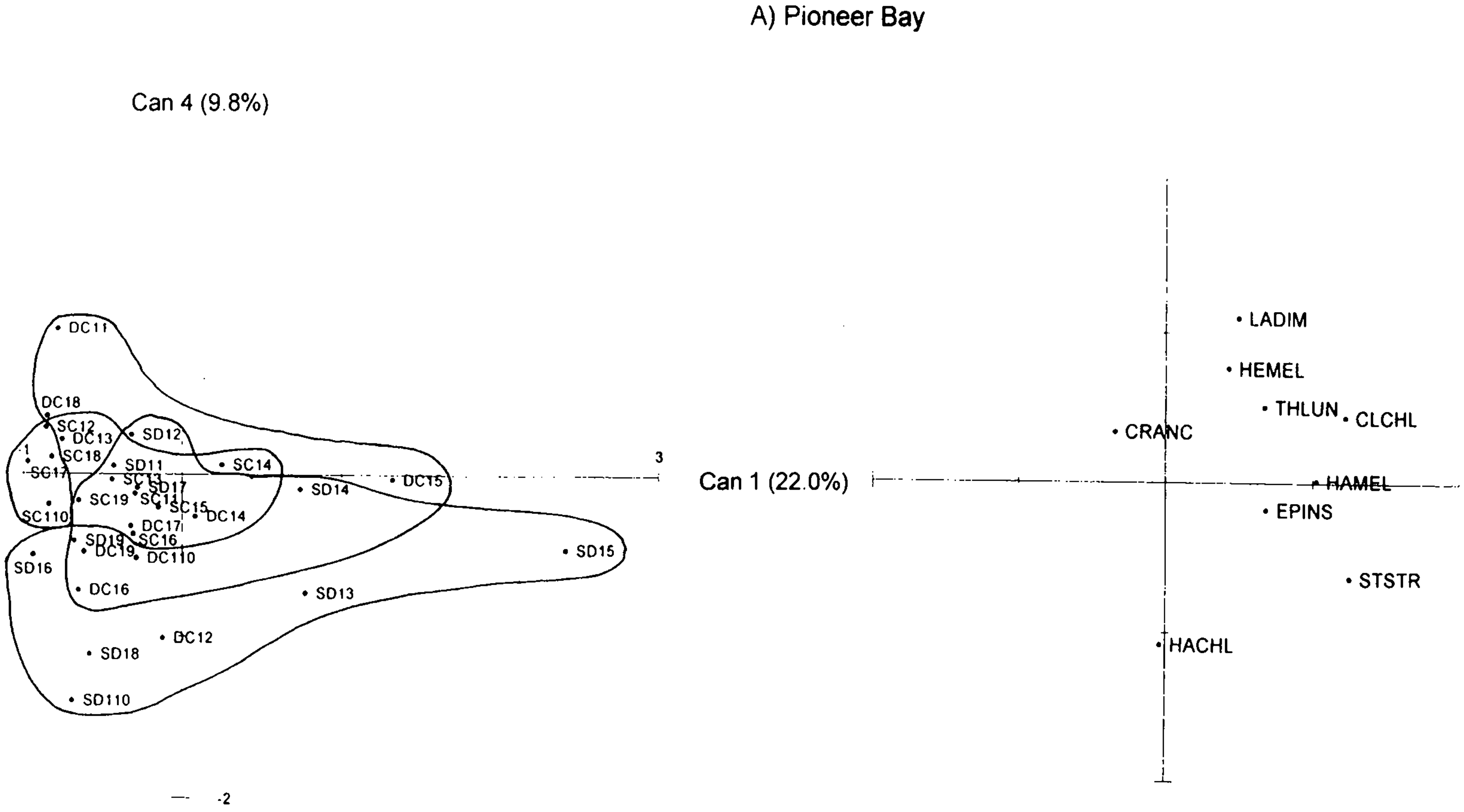


Fig. 6.14A. Ordination plots from CDA showing responses of the labrid assemblage to habitat manipulation in the soft coral degradation experiments at Pioneer Bay, Orpheus Island. Code names as in Fig. 6.13.

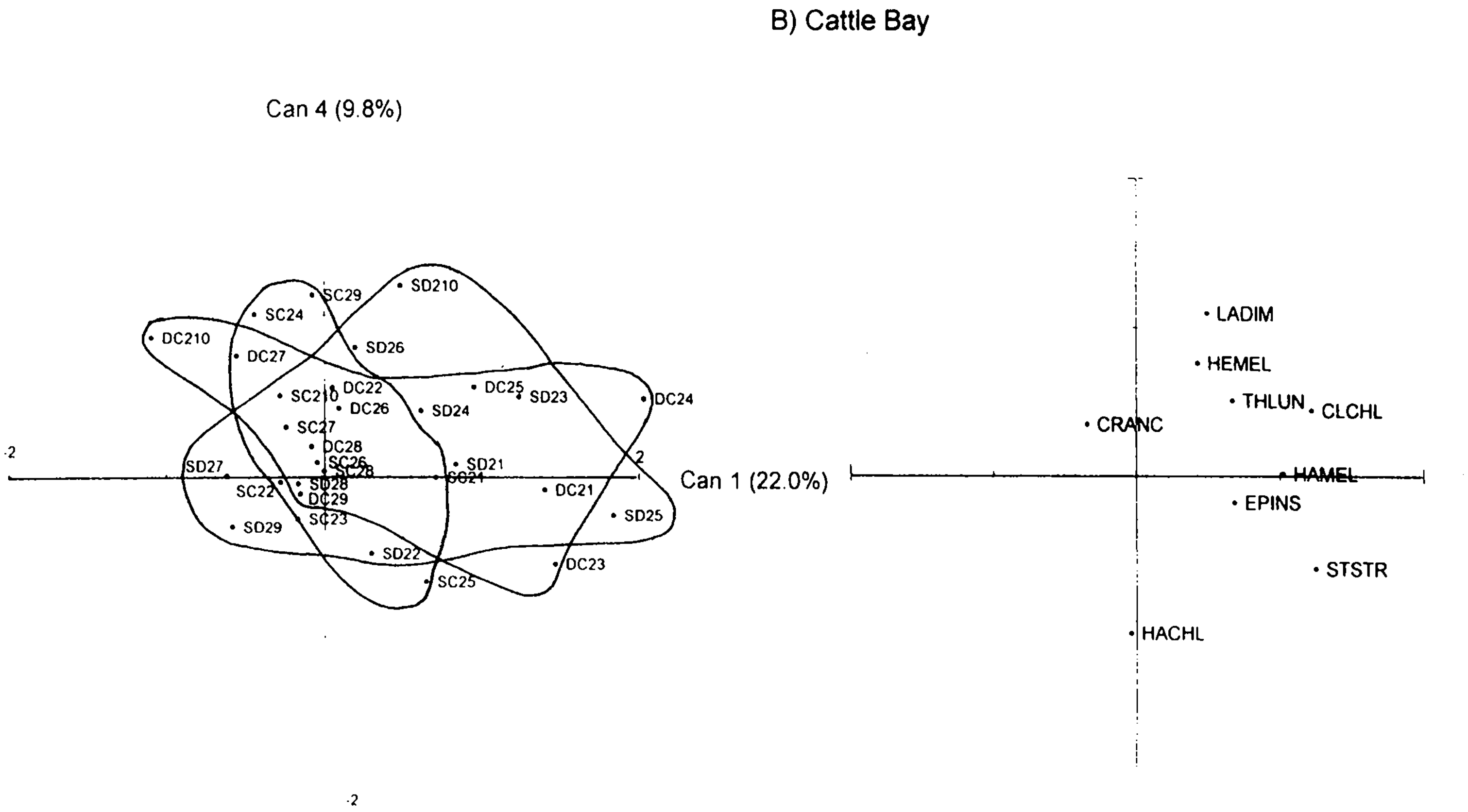


Fig. 6.14B. Ordination plots from CDA showing responses of the labrid assemblage to habitat manipulation in the soft coral degradation experiments at Cattle Bay, Orpheus Island. Code names as in Fig. 6.13.

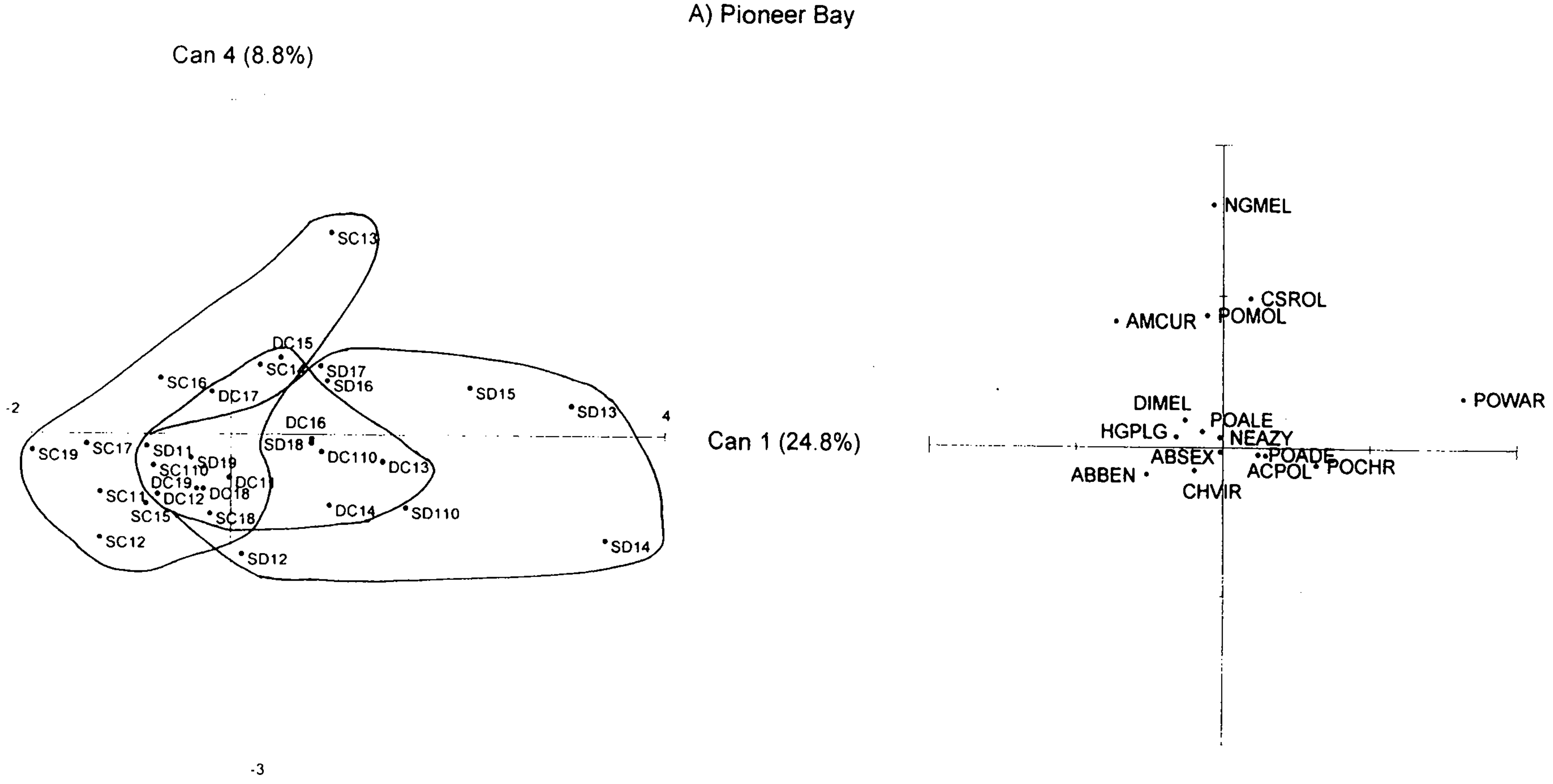


Fig. 6.15A. Ordination plots from CDA showing responses of the pomacentrid assemblage to habitat manipulation in the soft coral degradation experiment at Pioneer Bay, Orpheus Island. Code names as in Fig. 6.13.

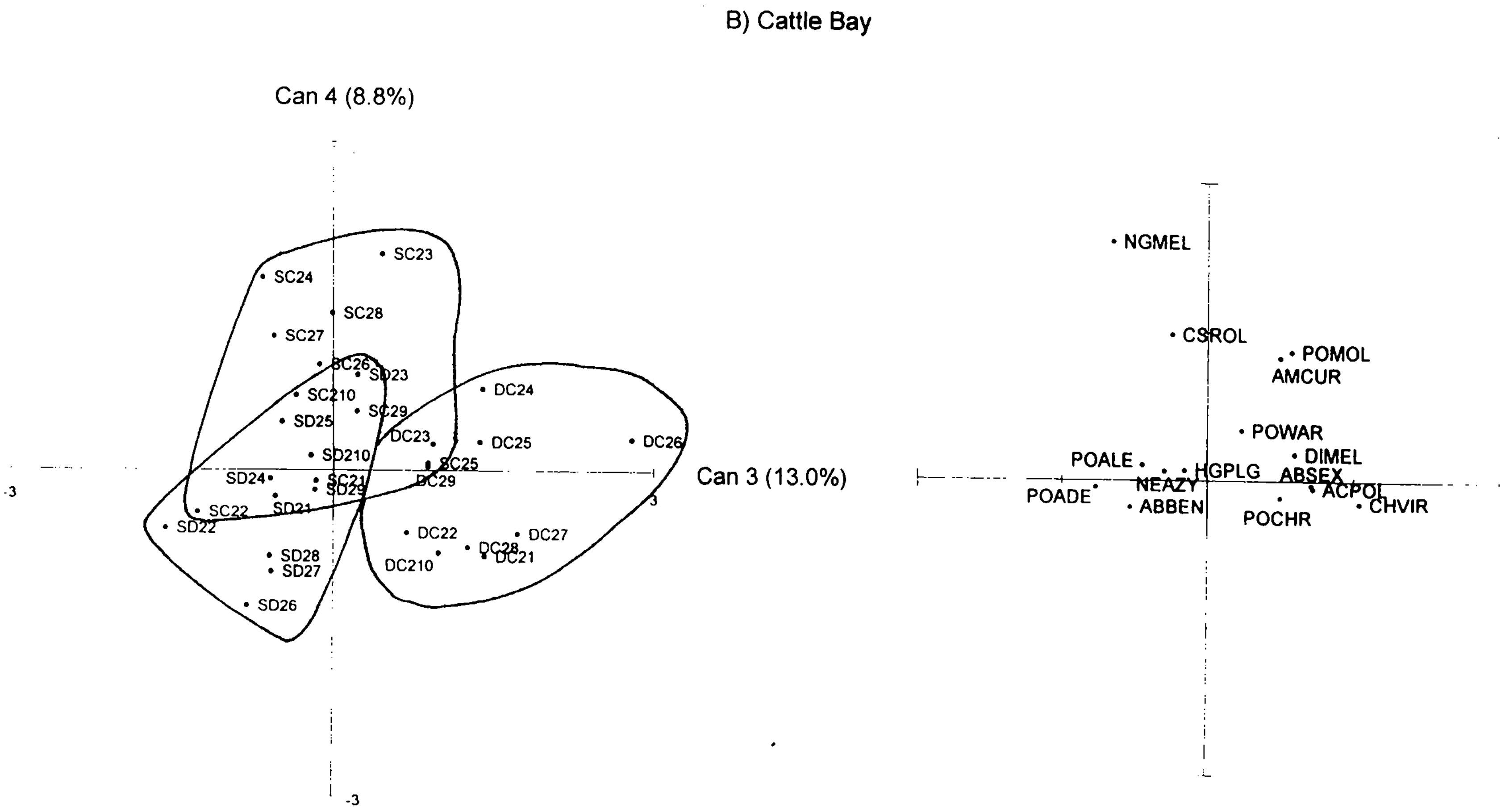


Fig. 6.15B. Ordination plots from CDA showing responses of the pomacentrid assemblage to habitat manipulation in the soft coral degradation experiment at Cattle Bay, Orpheus Island. Code names as in Fig. 6.13.

# CHAPTER 7

# EXPERIMENTAL EVALUATION OF THE EFFECT OF LOCAL HABITAT REHABILITATION ON CORAL REEF FISH ASSEMBLAGES AT TWO GEOGRAPHIC LOCATIONS

## 7.1 SUMMARY

Rehabilitation of coral reef habitat usually involves transplanting coral fragments or whole colonies to increase coral cover, but the flow on effects to local fish assemblages is unknown. Here I test the hypothesis that habitat rehabilitation is sufficient to increase the abundance and diversity of fishes associated with living coral reef. Experimental increases in the cover of three coral types (Branching *Acropora*, Massive and Soft corals) in replicated 25m$^2$ plots of degraded reef habitat were carried out at two geographic locations (Maiton Island, Phuket, Thailand; Orpheus Island, central GBR, Australia), and two sites within each of these locations. These three coral types dominate different patches of reef at these locations, and experimental increases were accomplished by transplanting coral from less disturbed sites. Greatest increases in diversity, species richness and overall abundance, and greatest changes in community structure were associated with the branching *Acropora* rehabilitation, although introduction of massive corals and soft corals also had effects. Assemblage structure tended to convulse on undisturbed fish assemblage associated with these substrata. The high complexity associated with branching *Acropora* forms is likely to be an important factor. There was some variation of the results between regions and between localities within each region, the differences reflecting the local reef conditions and the regional species pool. Results suggest that on a local scale, habitat rehabilitation may be sufficient to restore fish assemblages, provided there is a source of migrants or recruits.

## 7.2 INTRODUCTION

There is a widespread perception that coral reefs in many regions are in a state of decline (Brown, 1987; Wilkinson *et al.*, 1993; Eakin *et al.*, 1996). A combination of natural disturbances (cyclones, disease) and anthropogenic factors (eutrophication, sedimentation and overfishing) have all been implicated (Brown, 1987; Tomascik & Sander, 1987; Sorokin, 1993). Long-term declines in coral cover are usually associated with an increase algae, and this may represent a permanent habitat shift, particularly if processes impacting on the environment continue (Done, 1992; Hughes, 1994). Natural recovery may be slow or absent even in the absence of continued disturbance, if recruitment of coral and associated organisms is inhibited (Pearson, 1981). This situation raises the issue of pro-active measures to restore habitats. Restoration ecology is a relative new branch of ecology with few well-established principles (Simberloff, 1988; Jordan, 1995). However, experimental studies on marine habitat-forming organisms (saltmarshes, mangroves, corals) are providing information on appropriate methods and are helping resolve fundamental ecological issues about the roles of these organisms. While some attempts at restoring local areas of coral reefs have been undertaken (Bouchon *et al.*, 1981; Yap & Gomez, 1984, 1985; Harriot & Fisk, 1988a, 1988b; Hudson & Diaz, 1988; Yap *et al.*, 1992; Clark & Edwards, 1995), whether or not coral transplantation and establishment will lead to the restoration of fish assemblages is unknown.

Rehabilitation attempts directed at fish assemblages will rely heavily on information on fish-coral relationships, and rehabilitation experiments may themselves help resolve ecological issues relating to these interactions. The influence of structural factors on the composition of coral reef fish assemblages are not well understood. Historically, ecological studies have focused on demographic processes such as recruitment (Smith & Tyler, 1972; Sale, 1977; Talbot *et al.*, 1978; Victor, 1983; Doherty & Williams, 1988) or particular processes such as competition or predation (Hixon & Beets, 1989, 1993; Hixon & Menge, 1991). Habitat structure may play an important role in mediating all of these processes (Jones 1988, 1991; Hixon & Beets, 1989, 1993). There is increasing evidence of strong habitat selection at settlement (Sale, *et al.*, 1984; Eckert 1985a, b; Booth & Beretta, 1994; Green, 1994; Doherty, 1996) and that the presence of habitat refuges reduces the impact of predation on mortality (Jones, 1988; Hixon & Menge, 1991). Thus, habitat degradation may affect populations and communities in a variety of ways. Local habitat rehabilitation or enhancement represents a powerful tool for examining the mechanisms of fish-habitat associations.

The evidence that local habitat degradation has a detrimental effect on fish assemblages is increasing, both from observational (Sano *et al.*, 1984, 1987; Williams, 1986; Dawson-Shepherd *et al.*, 1992) and experimental studies (Sano *et al.* 1984). Declining habitat availability appears to lead to a decline in many fish species, suggesting that local populations may be resource limited. However, corresponding increases in algal habitat following coral decline do not appear to lead to corresponding increases in some herbivorous fishes, as might be expected (Wellington & Victor, 1985). It is possible that the responses of organisms to resource changes are asymmetrical. That is, patterns observed when resources decrease may not be reversed when resources are later increased. The potential for such asymmetries has not been previously been the subject of controlled experiments.

In this thesis, studies repeated at two widely separated geographic locations (Phuket, Thailand - Chapter 4) and (Orpheus Island, GBR, Australia - Chapter 5) confirmed that particular species or groups of fish have a predictable spatial relationship with different types of habitat structure. Patterns of abundance of some species were positively correlated with live coral cover, while others were negatively correlated. There could be regular patterns in the species richness of certain groups in relation to coral cover. Experimental reductions in coral cover induced patterns that establish that, at least for some locations, levels of coral play a causative role in influencing community structure and limiting particular populations.

In this chapter, I test the hypothesis that by experimentally increasing particular types and growth forms of coral in degraded reef areas, the effects of local degradation can be reversed. That is, the recovery of local populations and assemblages is limited by the recovery of appropriate habitats. As the effect may be scale dependent, the experiments are repeated on the same spatial scale as the disturbance experiments, and at the same two geographic regions and locations as in (Chapter 6).

## 7.3 MATERIALS AND METHODS

### 7.3.1 STUDY AREA

Similar experiments to test the effects of increasing coral cover were conducted at two geographic regions, Phuket (Thailand) and central Great Barrier Reef (Australia).

At Phuket, the field experiment was carried out at two localities (Northeast and Southeast reefs) on Maiton Island (7° 45′ N, 98° 29′ E ) 8 km south east of Phuket (Fig 7.1A). On the central GBR, the field experiment was carried out at two localities (Cattle and Pioneer Bays) on Orpheus Island (18° 35′ S, 146° 29′ E ) 16 km offshore (Fig. 7.1B). These study sites have been described in detail elsewhere (Chapter 6). Experiments were set up in relatively degraded areas at each location. At Maiton Island, the experiment was conducted at the depth of 4 to 5 m below mean sea level, in the middle of a very gradual reef slope. At Orpheus Island, experimental units were set up at 3 to 4 m below mean sea level at reef edge.

## 7.3.2 EXPERIMENTAL DESIGN

### 7.3.2.1 Experimental unit

Experiments were designed to test the effect of increasing the cover of particular types/growth forms of coral on the diversity and species-composition of fishes. All experimental increases were carried in standard 5x5 $m^2$ quadrats, which were monitored along with control plots of the same size. To standardise starting conditions, benthic cover of major coral types within each experimental unit was assessed in the field prior to allocation of treatments using a 1 $m^2$ quadrat with grid intersections at every 25 cm. Due to geographic differences in habitat structure, the experimental design differed slightly between the two locations.

### 7.3.2.2 Treatments

**Experiment 1: Maiton Island, Phuket.** Testing the effects of habitat rehabilitation at Maiton Island involved four treatments, with three replicates per treatment. The treatments were as follows:

Treatment 1: Dead coral control (DC) is a degraded reef area where recent living coral cover of each experimental unit is less than 5%.

Treatments 2 to 4 were initially the same as treatment 1 but were manipulated by translocation of different types.

Treatment 2: Branching *Acropora* coral rehabilitation (AR). *Acropora* colonies of approximately 30-50 cm. in diameter were translocated and randomly arranged to cover approximately 20% of the total unit area. To do this, corals were placed on a 1 $m^2$ quadrat first, to measure the area before re-arranging within the experimental unit. Colonies were attached by fixing to a steel rod, 0.7 cm. in diameter and 40 cm. long, which served as an anchor. All branching *Acropora* (*Acropora formosa*) colonies were transplanted from Aoe Island, 8 km west from Maiton Island. *Acropora formosa* was selected because it historically dominated the area, but has since been destroyed completely by storm. Dead coral fragments of this species were observed as coral debris throughout the area. The technique employed to transfer *A. formosa* was by breaking segments of *A. formosa* away from larger patches. Each coral unit was wrapped by news paper and sprayed with sea water to maintain high moisture levels and low temperature during the transportation. The duration of translocation (from collecting to experiment setting up) was between two and four hours. After one week, more than 90% of the translocated coral was still alive.

Treatment 3: Massive coral rehabilitation (MR) using *Porites lutea* colonies of approximately 0.5 - 1.5 m in diameter from surrounding area were translocated into the experimental unit and randomly arranged to cover approximately 20% of total unit area used the same technique as treatment 2.

Treatment 4: Soft coral rehabilitation (SR). *Nepthya* spp. usually attached to dead branching corals of approximately 30 - 50 cm. in diameter were translocated and randomly arranged to cover approximately 20% of the total unit area.

**Experiment 2: Orpheus Island, GBR.** For Orpheus Island, only three treatments, the dead coral control (DC), the Acropora rehabilitation (AR) and the soft coral rehabilitation (SR) were carried out in essentially the same way, with four replicates per treatment. Translocation of massive corals was not feasible in this area. *Acropora* colonies were translocated from nearby areas, with most coming from Cattle Bay, due to the low abundance of *Acropora* at Pioneer Bay. In the soft coral treatment, *Sinularia* spp. were used, which were usually attached to dead massive corals approximately 20 - 50 cm. in diameter.

### 7.3.3 DATA COLLECTION

The abundance of all fish species was assessed using a visual census technique with each plot monitored for 10-15min to record all individuals present. Cryptic and small species were excluded from the study, e.g. Gobiidae and Blenniidae.

The timing and duration of the experiments in the two regions differed. The experiments at Maiton Island were conducted between April 1994 to April 1995. Fish assemblage data were collected five times: 1) before habitat manipulation , 2) one week after manipulation, 3) four months after manipulation, 4) seven months after manipulation, and 5) ten months after manipulation. However, at time 3 (4 months after manipulation) data were collected only at NE-reef because of bad weather.

At Orpheus Island, data were collected during October 1995 to January 1996 for 10 times: 1) two weeks before habitat manipulation, 2) two weeks after manipulation, 3 to 5) every month and 6) to 10) every two months.

### 7.3.4 DATA ANALYSIS

Community level parameters including, species richness, species diversity, and overall abundance of each experimental unit were calculated. Data for each region were examined in separate analyses. Diversity was calculated using the Shannon-Wiener diversity index (H') which is calculated based on natural log though out (Pielou, 1974). Initially, the focus was on all species, followed by breaking these up into three groups/families, Labridae and Pomacentridae, and "Other" major families. However, the "Other" category was not considered for experiment at Maiton Island due to very low abundance and occurrence.

Separate analyses of variance were carried out for each locality of each region. For simplicity, two analysis were carried out for each locality, one the base-line "Before" sampling, to assess any pre-existing differences among treatments, and one based on all sampling "After" manipulation, to test for the effects of rehabilitation. the before and after habitat manipulation. A one way ANOVA was carried out for before manipulation and two way type I ANOVA (Factors: Treatment, Time, Treatment x Time) was carried out for after manipulation for each variable examined. A multi-factorial ANOVA was not used because the lack of power. *A priori* tests were carried out for species richness and overall abundance to satisfy the assumption of homogeneity of variances. Only overall abundance data was square-root transformed. Student-

Newman-Keuls (SNK) test was used as *a posteriori* test to identify the source of significant differences contributing to main effects.

Canonical Discriminant Analysis (CDA) was used to test for significant differences among treatments at each locality at each time based, on the distribution and abundance of fish in experimental unit. Data reduction was carried out before performing the CDA by using species present more than 10% of total units and times. Three groups/families: the Labridae, the Pomacentridae, and "Other" major families, were analysed separately, except at Maiton Island, where the "Other" category was of minor significance. *A priori* tests were carried out which lead to square-root transformation to satisfied the assumption of a multivariate normal distribution. Transformation was also used to reduce the influence of dominant species on an overall analysis.

Data from each region were analysed separately, but both localities of each region and all sampling times were analysed together. CDA was performed using square-root transformed and centred data matrix. The analyses were conducted using Proc CANDISC in SAS (SAS Institute Inc., 1990). The species responsible for the discriminant were considered from total canonical structure (correlation) because fish were considered as respond to treatment rather than influence (predict) the treatment.

## 7.4 RESULTS

### 7.4.1 REGIONAL PATTERNS IN COMMUNITY STRUCTURE

At Maiton Island, a total of 54 species (3,153 individuals) were recorded on experimental plots over the course of the experiment, including 25 labrids and 18 pomacentrids (Appendix 7.1). These two families dominated the assemblage. with labrids accounting for 50.6% and the pomacentrids 47.4% of the individuals recorded. Twelve labrid species and 8 pomacentrids occurred in more than 10% of all experimental units and times (Appendix 7.2A). The most common labrids were *Thalassoma lunare, Halichoeres hortulanus, Coris batuensis, H. vrolikii, H. timorensis, H. scapularis, Stethojulis interrupta, H. marginatus, H. argus, Coris pictoides, Cheilinus chlorourus, H. nebulosus, Labrichthys unilineatus*, and *S. trilineata* and the pomacentrids were dominated by *Pomacentrus chrysurus, P. adelus, P. similis, P.*

*moluccensis, Dascyllus aruanus, Chromis weberi, Plectroglyphidodon lacrymatus*, and *P. amboinensis*.

At Orpheus Island, a total of 59 species (18,061 individuals), including 20 labrids and 22 pomacentrids were recorded from habitat rehabilitation experiment, during the 10 times between October 1994 to December 1995. Here, pomacentrids accounted for 75.7% of the total abundance, with labrids the second most abundant (18.1%). Only 15 species of pomacentrids and nine labrids were present in more than 10% of all experimental units and times (Appendix 7.2B). Common species here for pomacentrids included *Pomacentrus moluccensis, P. wardi, P. adelus, Neoglyphidodon melas, Amblyglyphidodon curacao, Acanthochromis polyacanthus, Dischistodus melanotus, Abudefduf sexfasciatus, A. bengalensis, Hemiglyphidodon plagiometopon, Chromis viridis, P. alexanderae, Neopomacentrus azysron, P. chrysurus*, and *D. pseudochrysopoecilus* and the labrids included *Halichoeres melanurus, Thalassoma lunare, Stethojulis strigiventer, Labroides dimidiatus, Hemigymnus melapterus, H. chloropterus, Choerodon anchorago, Epibulus insidiator*, and *Cheilinus chlorourus*. The different biogeographic species pool between these two regions has already been mentioned in the previous chapter (6).

### 7.4.2 IMPACT OF REHABILITATION ON COMMUNITY PARAMETERS

The community parameters, including species richness, Shannon-Wiener diversity index and total number of individuals sampled were not strongly influenced by the rehabilitation experiment at Maiton island (Experiment 1). Few significant effects were detected by ANOVA (Table 7.1). There was a general trend towards increased diversity, species richness and abundance at both sites over the course of the experiment (Fig's 7.2-7.4). The effects of the rehabilitation that were detected differed for the two locations. Before manipulation, there were no significant differences among treatments at NE-Reef, but the experimental manipulation induced effects on the diversity and total abundance of labrids only. At this locality, there was considerable temporal variation in diversity (Fig. 7.2), species richness (Fig. 7.3) and total abundance (Fig. 7.4), and the treatments effects that were detected were not strong. Introduction of massive corals appeared to reduce the diversity of labrids (Fig. 7.2) and the effects of rehabilitation on the total abundance of labrids were fairly specific to each coral type (Fig. 7.4). The abundance of labrids in the soft coral treatment was not significantly different from the control. There were trends toward greater diversity and species richness of pomacentrids in the hard coral treatments at NE-Reef. The *Acropora* rehabilitation consistently recorded the highest mean diversity, species richness and total abundance for all taxonomic groupings.

There were also few effects of the coral rehabilitation on species diversity and abundance parameters at SE-Reef, Maiton Island (Table 7.1). The effects observed were complicated by chance differences among the treatments prior to the manipulation (Table 7.1). There were no significant effects (only trend) of the treatments on the diversity (Fig. 7.2), species richness (Fig. 7.3) or abundance (Fig. 7.4) of labrids.

Plots for the soft coral rehabilitation started out with greater pomacentrid diversity, species richness and abundance, with no significant difference among the other three treatments (Table 7.1). A treatment effect on pomacentrid (and overall) diversity, species richness and abundance appeared to be due to an increase in the massive coral treatment and the soft coral treatment, compared with the control (Fig's 7.2-7.4). The difference between the soft coral treatment and the control was greater following the manipulation, suggesting rehabilitation of this coral enhanced pomacentrid diversity and abundance. There were no effects of the *Acropora* rehabilitation.

The rehabilitation experiment at Orpheus Island had a much more substantial effect on the fish diversity and abundance, primarily due to the *Acropora* rehabilitation (Table 7.2). This treatment induced in major increase in abundance (Fig. 7.5), diversity (Fig. 7.6) and species richness (Fig. 7.7) at both locations, but particularly at Cattle Bay. Here, there were no significant differences among treatments prior to the experiment, but afterwards, the diversity, species richness and abundance of both labrids and pomacentrids (hence all species combined) were generally highest in the *Acropora* treatment. At Pioneer Bay, the effect was restricted to the Pomacentridae, the dominant component of the fish fauna at Orpheus Island. Overall, the soft coral enhancement had little effect and could not be distinguished from controls, although the abundance of pomacentrids (and all fish combined) at Cattle Bay was reduced by soft coral . There was a trend toward greater diversity and species richness of pomacentrids in the soft coral at Pioneer bay and the opposite effect at Cattle Bay. There were some pre-existing differences among treatments for the "Other" fish families, but no effects of the manipulation. In general, these community parameters showed considerable temporal variation that was independent of the treatments.

### 7.4.3 IMPACT OF REHABILITATION ON SPECIES COMPOSITION

Canonical discriminant analysis of the experiment at Maiton Island exposed treatment effects and other trends in the species composition of labrids (Table 7.3). The first four canonical discriminants were significant, accounting for 70.9% of the variation. Canonical discriminants 2 and 3 (explaining 28.8%) best described the influence of rehabilitation on labrid species composition at NE-Reef, while discriminants 1 and 3 (45%) described the treatment effect for SE-Reef (Fig. 7.8). At NE-Reef, a clear influence of the treatments was evident, especially at times 4 and 5, when the treatments were maximally contrasted (Fig. 7.8A). The species responsible for these patterns were as follows: In the dead coral control (DC), the abundance of *Coris pictoides* increased towards the end of the experiment (times 4 and 5). The abundance of *Halichoeres vrolikii* and *Coris batuensis* increased in the *Acropora* rehabilitation (AR) after time 3 (Appendix 7.3). More *Thalassoma lunare* were found in massive coral rehabilitation (MR) after manipulation (Fig. 7.8A) especially at time 5. For the soft coral rehabilitation (SR), some differences in community structure arose at time 3 and time 4, but at the end of the experiment, the labrid assemblage had returned to a composition similar to base-line conditions (Fig. 7.8A).

At SE-Reef, there was greater temporal variability in the species composition of labrid assemblages in the rehabilitation treatments compared with the controls, which remained relatively constant (Fig. 7.8B). The influence of massive coral and soft coral rehabilitation is evident from canonical discriminant 3, which is explained by higher abundance of *Stethojulis interrupta, Coris pictoides* and *Halichoeres scapularis* from time 3 to time 5. The *Acropora* rehabilitation had the greatest effect, with both canonical discriminants distinguishing it from the control, particularly at the end of the experiment. The species responding to *Acropora* rehabilitation were *Coris batuensis, H. scapularis* and *S. interrupta,* which increased in abundance relative to the control (Appendix 7.3).

For pomacentrids at Maiton Island, the first two canonical discriminants were significant and explained 72.3% of the variation. However, canonical discriminants 1 and 3 (55.7%) best illustrated the effect of rehabilitation for both locations (Fig. 7.9).

There was a strong effect of the *Acropora* rehabilitation at NE-Reef, particularly by the end of the experiment. The other treatments remained fairly similar to the controls and were relatively invariant. The main species responding to *Acropora* rehabilitation was *Dascyllus aruanus,* which increased in abundance after the addition of the branching coral (Appendix 7.3). At SE-Reef, there was very little effect of the *Acropora* manipulation. Here, most the changes was induced by soft coral rehabilitation (Fig. 7.9B). There was also a small influence of the massive coral rehabilitation at both localities from time 3 onwards. The abundance of *Pomacentrus adelus* and *P. moluccensis* increased in the massive coral rehabilitation (Appendix 7.3).

At Orpheus Island, the results from CDA indicated that the first three canonical discriminants were significant, explaining 63.6% of the variation in labrid community structure. However, canonical discriminants 1 and 3 were used to illustrate the effects of rehabilitation at Pioneer Bay while canonical discriminants 1 and 4 were used for Cattle Bay (Fig. 7.10). At Cattle Bay, the effect of *Acropora* rehabilitation was evident from canonical discriminant 3. The *Acropora* and control treatments started off at the same point on this axis, but were always distinct after the addition of coral (Fig. 7.10A). The pattern suggests that there were no particular trends in community structure over time. There was little effect of the soft coral rehabilitation for labrids, which tracked changes in community composition that occurred in the controls.

At Cattle Bay, the effect of treatment was primarily based on canonical discriminant 4 (all treatments at time one lay in the same position). A short-term effect of soft coral rehabilitation can be seen at times 1 and 2 (Fig. 7.10B). By time 10, the labrid assemblages in all treatments had returned to near their initial composition. In general, little variation in labrid assemblages could be explained by the experimental treatments (Appendix 7.4).

For pomacentrids, the first three canonical discriminants (explaining 58.9% of the variation) were significant. The first two canonical discriminants (49.4%), indicated a strong effect of the treatments through time (Fig. 7.11). At Pioneer Bay, a treatment effect was clearly visible only for *Acropora* rehabilitation at time 9. Apart from that, all treatments appeared to exhibit a similar temporal trend. At Cattle Bay, treatment effect could be described primarily from canonical discriminant 1 (explaining 32% of the variation). The *Acropora* rehabilitation was maximally discriminated from the control at the end of the experiment. The pomacentrids responding to *Acropora* rehabilitation were *Amblyglyphidodon curacao, Acanthochromis polyacanthus*, and *Pomacentrus moluccensis*, which all showed increases in abundance relative to both the dead coral control and soft coral rehabilitation (Appendix 7.4).

Four species accounted for most of the variation in the "Other" major families. The results of the CDA indicated the first two canonical discriminants (70.4%) were significant. However, canonical discriminants 1 and 3 (65.1%) were considered for both localities (Fig. 7.12). The influence of *Acropora* rehabilitation and soft coral rehabilitation were detected as short term effects at times 3 and 6 at Pioneer Bay (Fig. 7.12A). All four species responded with an increase in their abundance. Long term effects, however, were not detected. At Cattle Bay, there were no obvious patterns in relation to the treatments (Fig. 7.12B).

## 7.5 DISCUSSION

The results suggest that local-scale habitat rehabilitation will also promote changes in and a recovery of fish assemblages, but there may be regional and local differences in the magnitude and the time scale of the response. In general, there were positive responses to re-introduction of particular coral types, with increases in the species richness, diversity and/or overall abundance of particular groups of fish, particularly site-attached pomacentrids, particularly at Orpheus Island. Most of the effects were attributable to species using coral as shelter rather than a food source, as coral feeding fish were rarely present in experimental areas. Positive effects of habitat rehabilitation and enhancement on community parameters were reported elsewhere (Clark & Edwards, 1994).

The effects of habitat rehabilitation were specific to the coral types that were translocated and particular families and species of reef fishes. The effects were consistent with patterns in community structure of fishes naturally associated with the different substrata. Branching *Acropora* had positive effects on the community parameters of all fish groups except "Other" families at Orpheus Island. At Maiton Island, the species responding with increased abundance to *Acropora* rehabilitation were *Halichoeres vrolikii, Coris batuensis,* and *Stethojulis interrupta.* For Pomacentrids, a substantial increase in abundance occurred for *Dascyllus aruanus*, a response that has also been reported by Sano *et al.*, (1984). A negative effect of *Acropora* rehabilitation can also be seen from sand-dwelling species, especially *Coris pictoides,* which recruited mainly on to controls and in low numbers on *Acropora* rehabilitation plots. At Orpheus Island, labrid species which responded with increased abundance to *Acropora* rehabilitation were *Halichoeres melanurus, Stethojulis strigiventer, Hemigymnus melapterus* at Pioneer Bay, and *Labroides dimidiatus* at Cattle Bay. For Pomacentrids, a substantial increase in abundance occurred for *Acanthochromis polyacanthus, Neopomacentrus azysron, Pomacentrus adelus* and *P. moluccensis.*

Massive coral rehabilitation only had an effect at Maiton Island where its influence on fish assemblage structure was less than for *Acropora.* The pomacentrids *Pomacentrus adelus, P. moluccensis* and *Chromis weberi* increased with the addition of massive corals, but the wrasses *Halichoeres vrolikii* and *Coris batuensis* decreased.

For soft coral rehabilitation, a positive response was detected only at NE-Reef of Maiton island where abundance of *H. vrolikii, C. batuensis* and *H. marginatus* increased. At Orpheus Island, labrid species responding with increased abundance to soft coral rehabilitation were *Thalassoma lunare* at both localities, *Halichoeres melanurus* at Pioneer Bay, and *Hemigymnus melapterus* at Cattle Bay. For Pomacentrids, an increase in abundance occurred only at Pioneer Bay for *Neopomacentrus azysron* and *Pomacentrus alexanderae.* A negative effect of soft coral rehabilitation was also seen on *Acanthochromis polyacanthus.* However, generally there were few effects of soft coral rehabilitation, reflecting the low use of this substratum by fishes. As was shown in Chapter 6, removal of soft coral can lead to an increase in abundance and diversity.

It was predicted that an increase in the cover of target life-forms from less than 5 to 20-25% should have positive effect, as the relationships between assemblages and habitat cover were always positive over this range (Chapters 4 and 5). However, the influence form *Acropora* rehabilitation generally had greater effects than massive and soft corals. This is probably because *Acropora* provides more complexity, with more effective shelter for fishes, compared with massive and soft corals. The increase in coral cover of only 20% may not have been sufficient to provide significant changes in fish assemblage structure for these corals. In contrast, Bell & Galzin (1984) reported positive response of fish assemblages to very small increases in of living coral over the

0 to 5% cover range. However, in correlative studies of this nature it is uncertain whether changes in fish assemblages are related to changes in coral cover, or other factors, such as reef zonation.

The family-specific nature of the relationships between fish and habitat structure may explain some differences in their response to habitat rehabilitation. For example, most of pomacentrids are extremely site-attached and closely associated with shelter (Sano *et al.*, 1987). There response to rehabilitation may be primarily through recruitment rather than movement, an so may occur over longer time scales than some other groups. Pomacentrids generally respond to changes in habitat structure in the long-term (Williams, 1986; Sano *et al.*, 1987). Effects on wrasses may be more immediate, through movement of individuals, but they may also be more transitory in their response as was the case for labrids at Maiton Island.

Differences in the response to habitat rehabilitation between regions may reflect the biogeographic peculiarities of fish-habitat interactions or regional differences in the condition of the reefs examined. The substantial effects on species diversity and abundance at Orpheus Island may be expected, given the dominance of pomacentrids on inshore reefs of the GBR. Alternatively, it may be attributable to reef status. Reefs at Maiton Island, Phuket were in a more degraded condition, so there were fewer sources of immigrants to colonise rehabilitated sites from adjacent areas. Greater responses by labrids here may reflect their increased importance in the fish assemblages in this region or their greater mobility. In contrast, reefs at Orpheus Island show no signs of serious degradation, and consequently have a more continuous habitat structure and more stable fish assemblages. The potential for rapid recovery due to the redistribution of fishes from other live coral areas is considerably greater. The temporary immigration of planktotrophic species such as *Abudefduf* spp., *Neopomacentrus* spp., and *Chromis* spp. were commonly observed at Orpheus Island.

Since the type of fish assemblage that develops is dependent upon the coral types that are restored, care must be taken to restore coral assemblages to the natural condition or range for an area. The degree of habitat rehabilitation required to restore fishes may vary, depending on the taxa involved and the status of the area. Small-scale pilot programs may be necessary to determine the optimal rehabilitation strategy for each area. It may take longer to restore local patches in areas with large-scale habitat damage, as sources of migrants or recruits may be scarce. In such cases, it may be necessary to actively restore fish assemblages by more direct means

Most of the previous studies of habitat enhancement have usually involved provision of artificial habitats adjacent to reefs. Rapid colonisation of artificial structures by reef fish have been reported in most studies and a diversity and abundance comparable to undisturbed natural reefs have often been achieved (Sano *et al.*, 1984; Ambrose & Swarbrick, 1989; Bohnsack *et al.*, 1994; Clark & Edward, 1994). Clark & Edward (1994) reported that community structure of reef fish found on the artificial habitat is usually very different from the nearby natural reefs. In this study, in contrast, found that reef fish assemblage on living coral rehabilitation treatment tend to have similar composition with correspond natural living coral area. Hence, construction of artificial reefs is not an alternative to rehabilitation of living coral habitats.

The need for habitat rehabilitation will vary depending on the type of disturbance (natural or human-induced), the scale of the disturbance and the potential for self- recovery. There is obviously greater potential for local self-recovery in places like Orpheus Island and much less scope for inducing large-scale changes. Habitat rehabilitation in such areas may be restricted to cosmetic changes in the vicinity of tourist resorts or pontoons. Habitat rehabilitation should primarily be conducted in seriously degraded areas with a low ability of self recovery. The kinds of experiments carried out here need to be scaled up to look at the potential of restoring much larger areas. However, without knowledge of historic conditions of a habitat, it can be difficult to gauge the success of a rehabilitation program.

The results of this study provide some of the first data to suggest fish assemblages will respond rapidly to rehabilitation of their habitat. In fact, fishes appear to so readily respond to habitat changes that care must be taken to restore appropriate habitats. While habitat modification may be used to target certain groups of fishes desirable as fisheries resources, habitat rehabilitation should be seen primarily as a conservation tool. The next step is to examine the effects of larger-scale attempts at rehabilitation and mechanisms that may accelerate the process. Coral transplantation may be used in conjunction with other techniques that may enhance recruitment of corals (e.g. algal turf removal) and coral reef fishes (e.g. light attraction devices - Munday *et al.*, in prep.). Research into rehabilitation should continue to help resolve fundamental ecological issues about processes limiting populations and structuring communities of reef fishes.

Table. 7.1. ANOVA and SNK test results of three community parameters (species diversity, species richness, total abundance) showing the response of coral reef fish assemblages to habitat changes in the habitat rehabilitation experiment at two localities surrounding Maiton Island, Phuket, Thailand. Before manipulation (Before) used a one way ANOVA while after manipulation (After) used a two way ANOVA of time and treatment effects (all fixed factors). DC = Dead coral control, AR = Acropora rehabilitation, MR = Massive coral rehabilitation and SR = Soft coral rehabilitation. Order of treatments (if presented) was from maximum to minimum value. * = significant at $P < 0.05$, ns = non-significant, underline bar indicated non-significant different (SNK) between treatments.

| Fish group | Community | NE-Reef | | SE-Reef | |
|---|---|---|---|---|---|
| | parameter | Before | After | Before | After |
| Overall | Diversity | treatment$^{s}$ | time* | treatment$^{s}$ | All factors $^{ns}$ |
| | Richness | treatment$^{s}$ | time* | treatment*<br>SR AR MR DC | treatment*<br>SR MR AR DC |
| | Abundance | treatment$^{s}$ | time* | treatment*<br>SR AR MR DC | treatment*<br>AR MR CR DC |
| Labridae | Diversity | treatment$^{s}$ | treatment*<br>AR SR DC MR | treatment$^{s}$ | All factors $^{ns}$ |
| | Richness | treatment$^{s}$ | All factors $^{ns}$ | treatment$^{s}$ | All factors $^{ns}$ |
| | Abundance | treatment$^{s}$ | treatment*<br>AR MR DC SR | treatment$^{s}$ | All factors $^{ns}$ |
| Pomacentridae | Diversity | treatment$^{s}$ | time* | treatment*<br>SR AR DC MR | time*, treatment*<br>SR MR DC AR |
| | Richness | treatment$^{s}$ | time* | Treatment*<br>SR AR DC MR | time*, treatment*<br>SR MR DC AR |
| | Abundance | treatment$^{s}$ | time* | Treatment*<br>SR AR DC MR | treatment*<br>SR MR AR DC |

Table. 7.2. ANOVA and LSD test results of three community parameters (species diversity, species richness, total abundance) showing the response of coral reef fish assemblages to habitat changes in the habitat rehabilitation experiment, at two localities surrounding Orpheus Island, central Great Barrier Reef, Australia. Before manipulation (Before) used a one way ANOVA while after manipulation (After) used a two way ANOVA of time and treatment effect (all fixed factors). DC = Dead coral control, AR = *Acropora* rehabilitation and SR = Soft coral rehabilitation. Order of treatments (if presented) was from maximum to minimum value. * = significant at $P < 0.05$, ns = non-significant, underline bar indicated non-significant different (SNK) between treatments.

| Fish group | Community parameter | Pioneer Bay |  | Cattle Bay |  |
|---|---|---|---|---|---|
|  |  | Before | After | Before | After |
| Overall | Diversity | treatment$^{s}$ | time* | treatment$^{s}$ | time*, treatment*<br>AR SR DC |
|  | Richness | treatment$^{s}$ | time*, treatment*<br>AR SR DC | treatment$^{s}$ | time*, treatment*<br>AR DC SR |
|  | Abundance | treatment$^{s}$ | time*, treatment*<br>AR SR DC | treatment$^{s}$ | treatment*<br>AR DC SR |
| Labridae | Diversity | treatment$^{s}$ | time* | treatment$^{s}$ | time*, treatment*<br>AR DC SR |
|  | Richness | treatment$^{s}$ | time* | treatment$^{s}$ | time*, treatment*<br>AR DC SR |
|  | Abundance | treatment$^{s}$ | time* | treatment$^{s}$ | time*, treatment*<br>AR DC SR |
| Pomacentridae | Diversity | treatment$^{s}$ | time*, treatment*<br>AR SR DC | treatment$^{s}$ | interaction*<br>AR DC SR |
|  | Richness | treatment$^{s}$ | treat*<br>AR SR DC | treatment$^{s}$ | time*, treatment*<br>AR DC SR |
|  | Abundance | treatment$^{s}$ | time*, treatment*<br>AR SR DC | treatment$^{s}$ | treatment*<br>AR DC SR |
| Other families | Diversity | treatment$^{s}$ | time* | treatment*<br>AR SR DC | time*, treatment*<br>AR SR DC |
|  | Richness | treatment$^{s}$ | time* | treatment*<br>AR SR DC | time*, treatment*<br>AR SR DC |
|  | Abundance | treatment$^{s}$ | All factors$^{ns}$ | treatment*<br>SR AR DC | time*, treatment*<br>AR SR DC |

Table 7.3. Results of multivariate significance tests for canonical discriminant analysis of the effect of different habitat manipulation (treatments), time and locality on different fish assemblages from two experiments carried out at two regions.

| Region | Fish assemblage | df | Pillai's trace | F | P |
|---|---|---|---|---|---|
| Maiton Island | Labridae | 420, 864 | 5.49 | 1.74 | <0.001 |
| | Pomacentridae | 280, 576 | 3.52 | 1.61 | <0.001 |
| Orpheus Island | Labridae | 531, 1620 | 3.11 | 1.61 | <0.001 |
| | Pomacentridae | 885, 2700 | 4.73 | 1.40 | <0.001 |
| | Other families | 236, 720 | 1.50 | 1.84 | <0.001 |

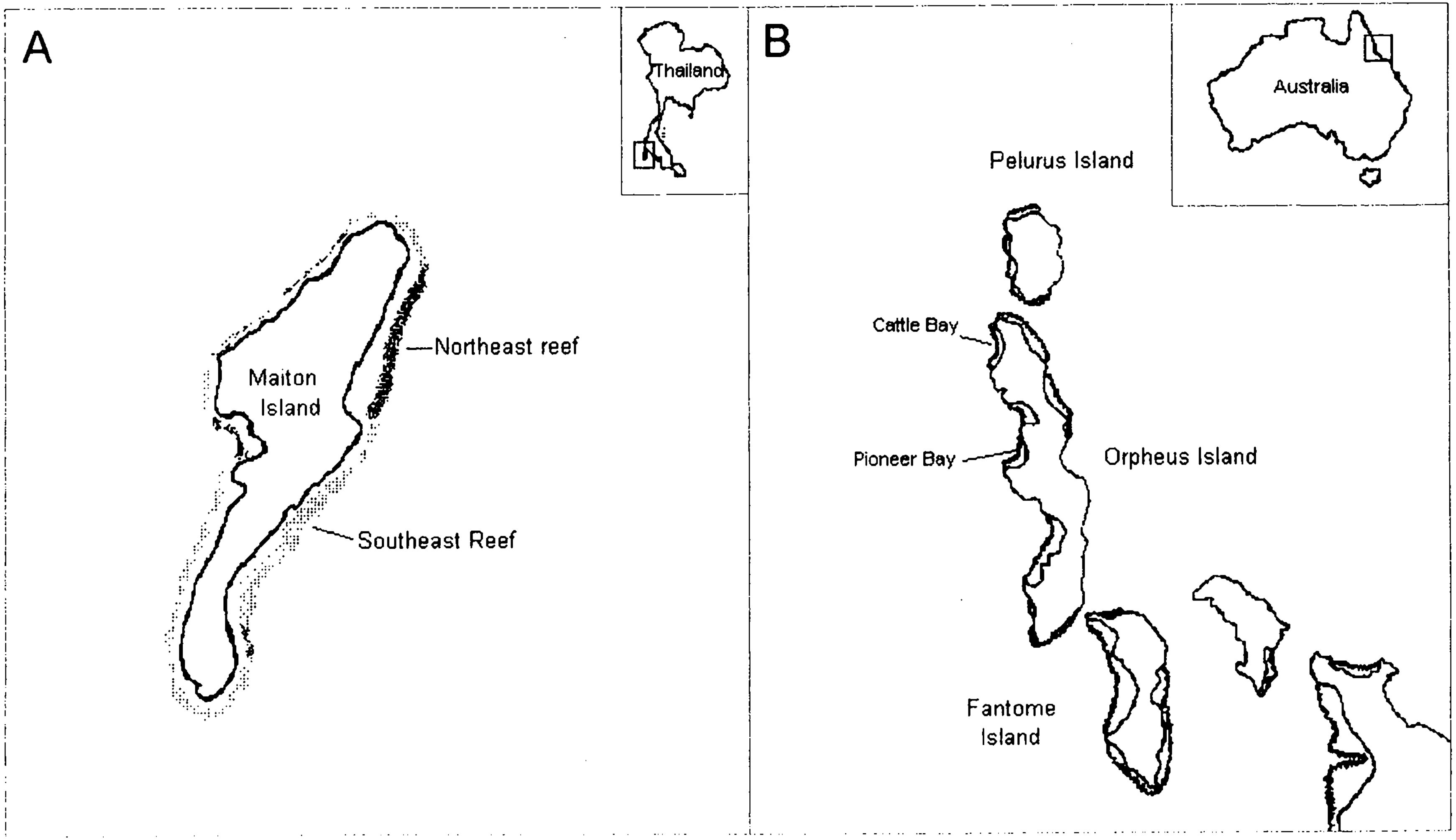


Fig. 7.1. Maps showing study areas, A) Maiton Island - Phuket, Thailand and B) Orpheus Island - Central Great Barrier Reef, Australia. In each region, two localities (Northeast reef, Southeast reef and Cattle Bay, Pioneer Bay respectively) were selected for conducting experiment.

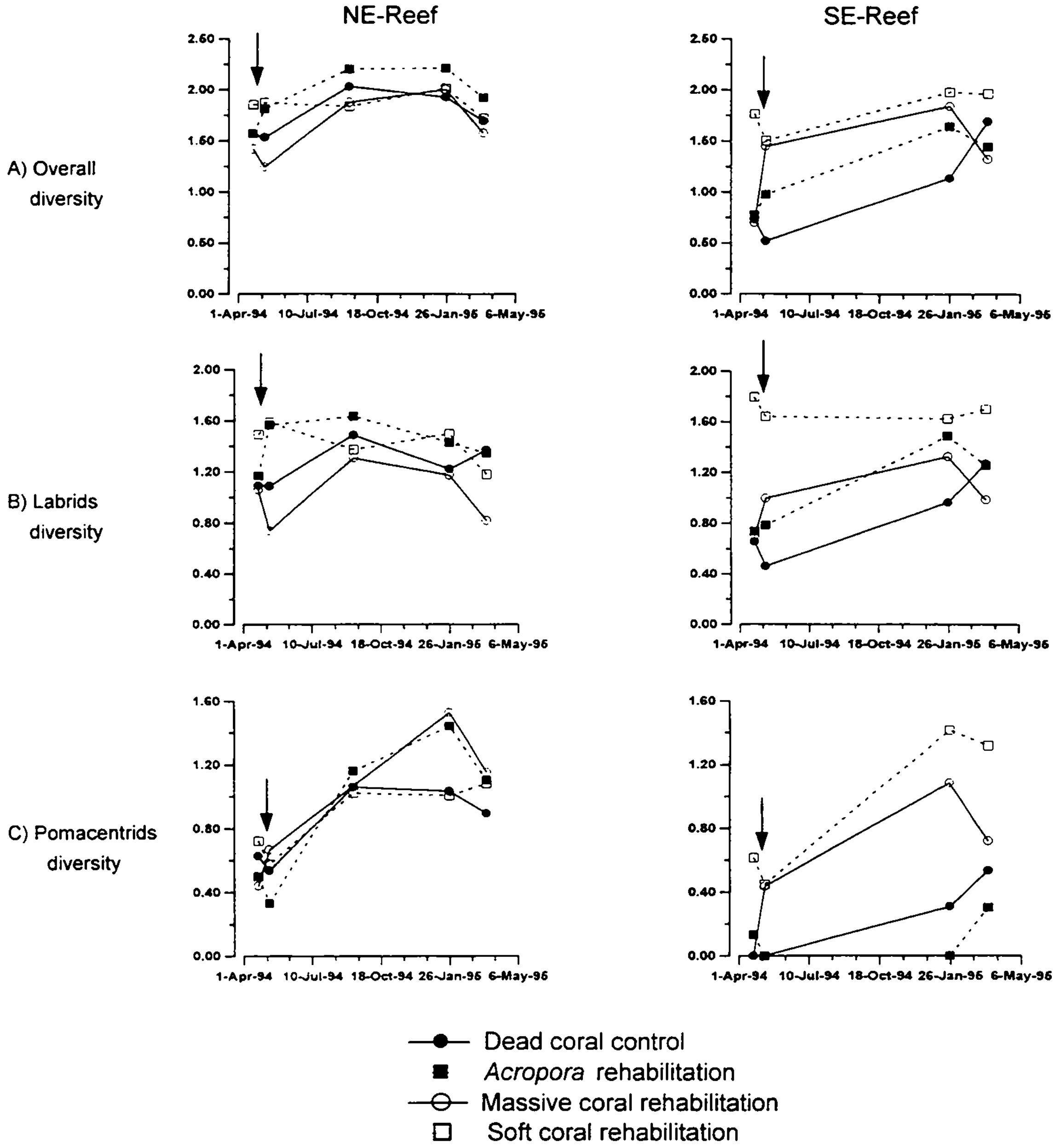


Fig. 7.2. Species diversity (mean diversity /25 sq. m) of A) Overall fish, B) Labrid, and C) Pomacentrid assemblages during habitat rehabilitation experiments at two localities on Maiton Island. Arrows indicate time of manipulation.

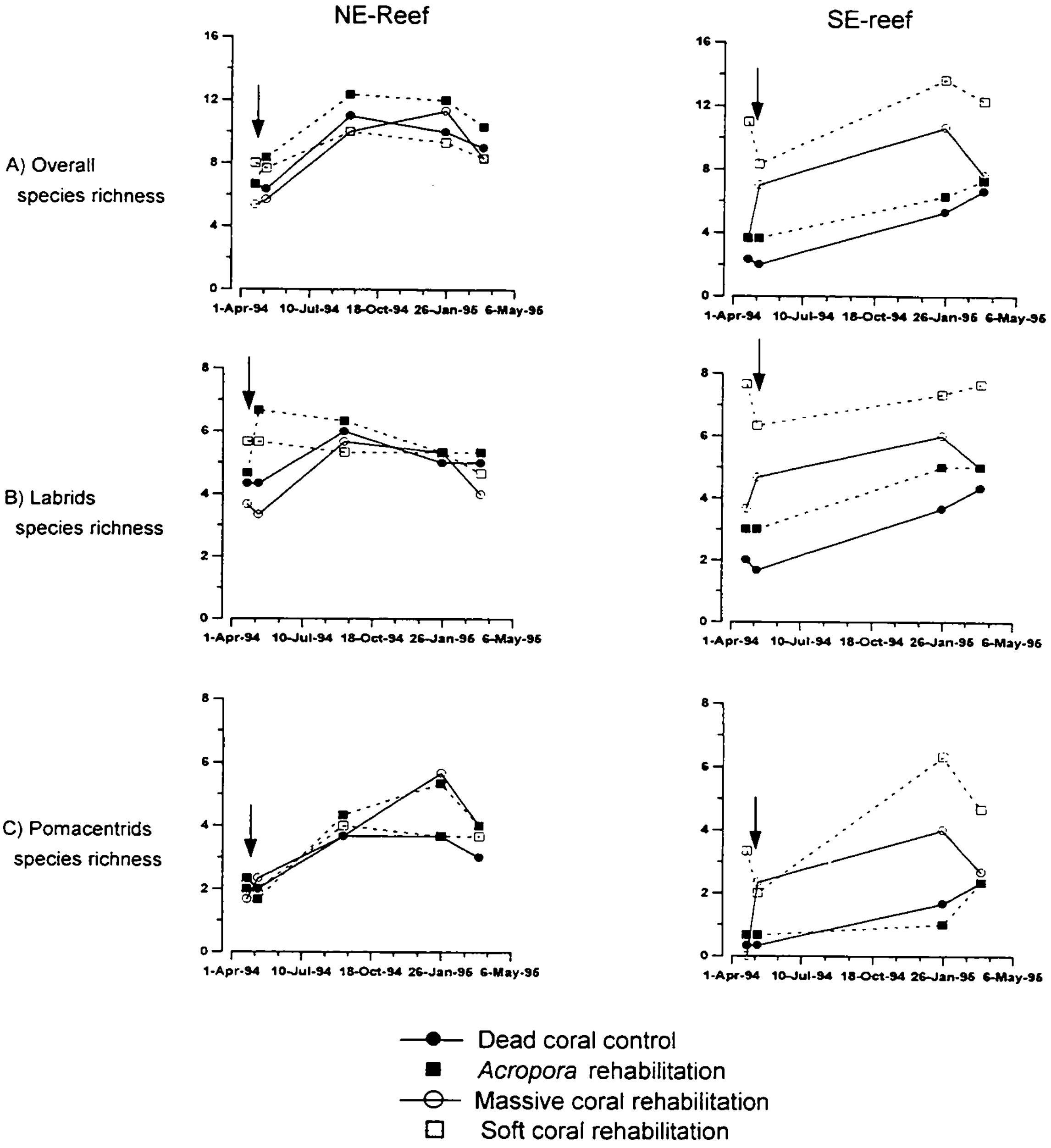


Fig. 7.3. Species richness (mean no. species/25 sq. m) of A) Overall fish, B) Labrid, and C) Pomacentrid assemblages during habitat rehabilitation experiments at two localities on Maiton Island. Arrows indicate time of manipulation.

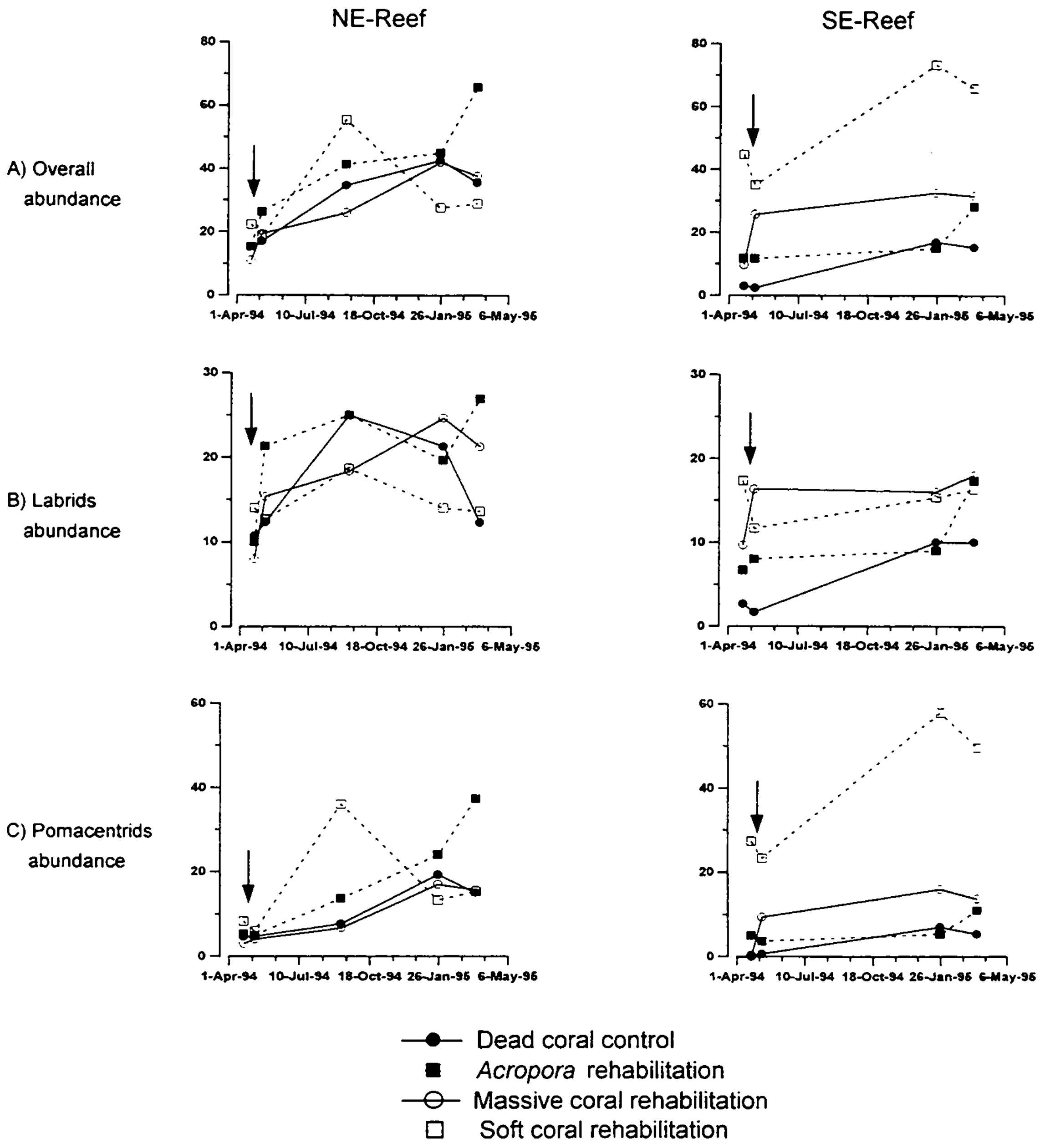


Fig. 7.4. Total abundance (mean no. fish/25 sq. m) of A) Overall fish, B) Labrid, and C) Pomacentrid assemblages during habitat rehabilitation experiments at two localities on Maiton Island. Arrows indicate time of manipulation.

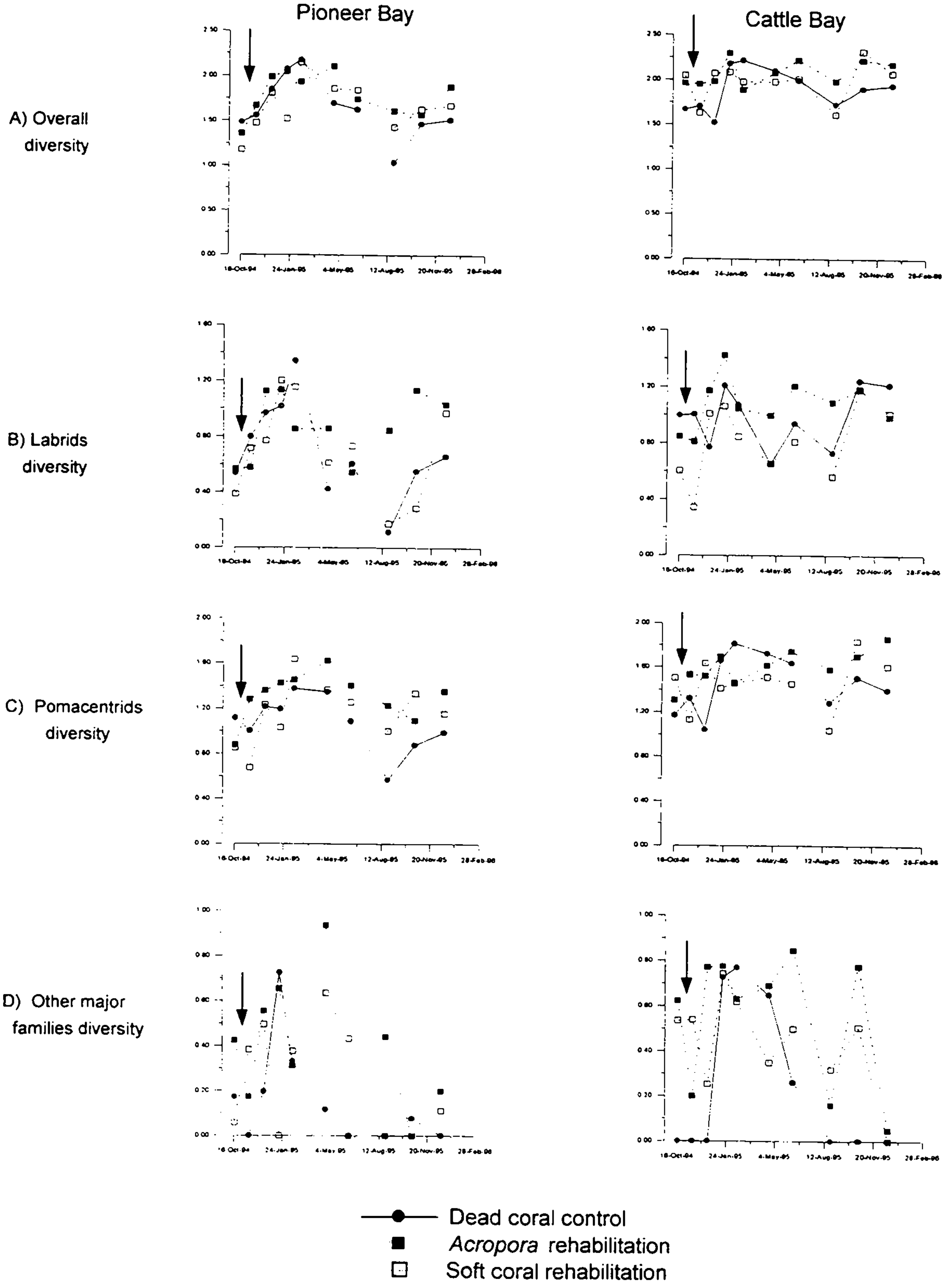


Fig. 7.5. Species diversity (mean diversity/25 sq. m) of A) Overall fish, B) Labrid, C) Pomacentrid, and D) Other major family assemblages for habitat rehabilitation experiments at two localities on Orpheus Island. Arrows indicate time of manipulation.

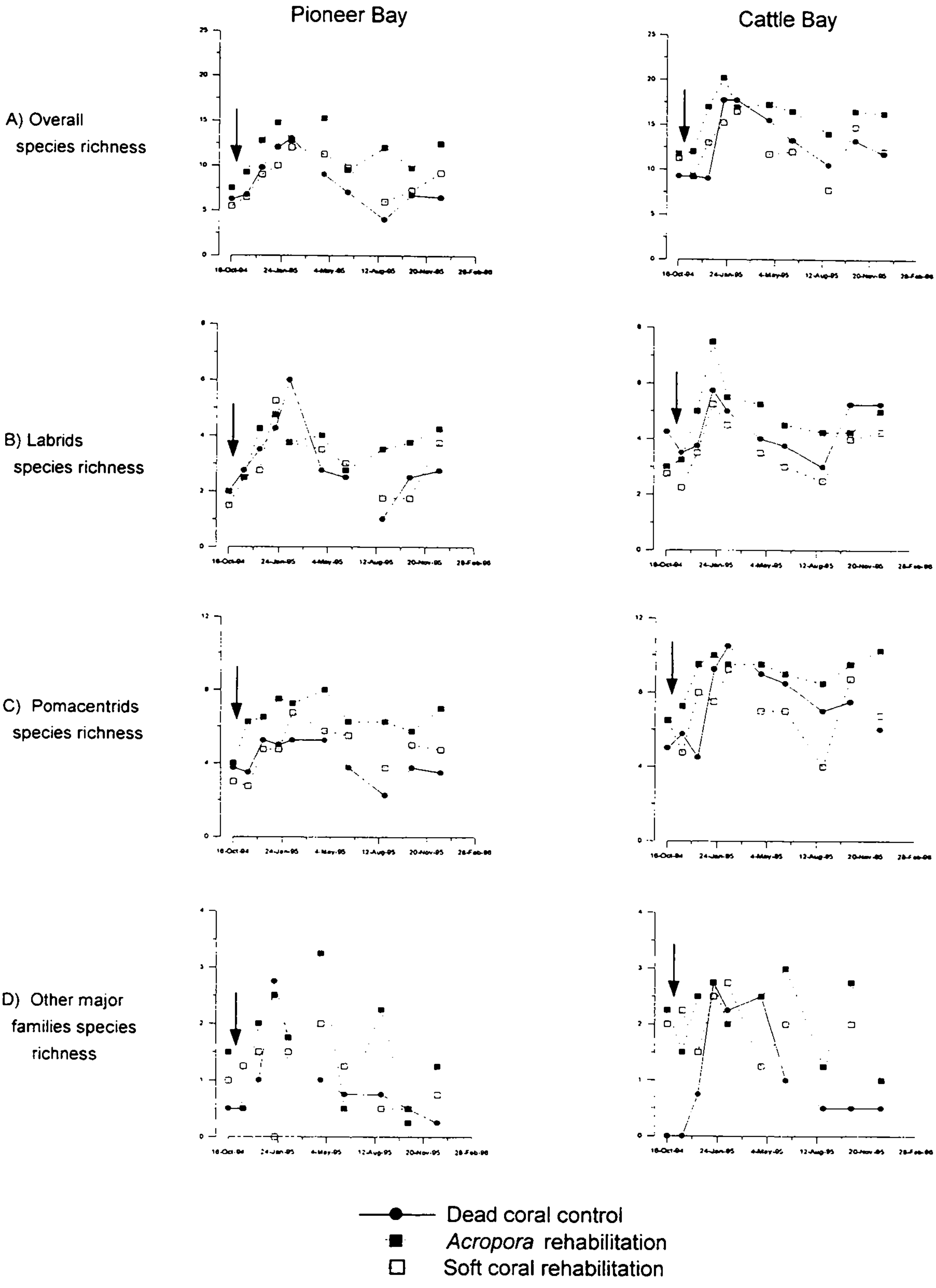


Fig. 7.6. Species richness (mean no. species/25 sq. m) of A) Overall fish, B) Labrid, C) Pomacentrid and D) Other major family assemblages during habitat rehabilitation experiments at two localities on Orpheus Island. Arrows indicate time of manipulation.

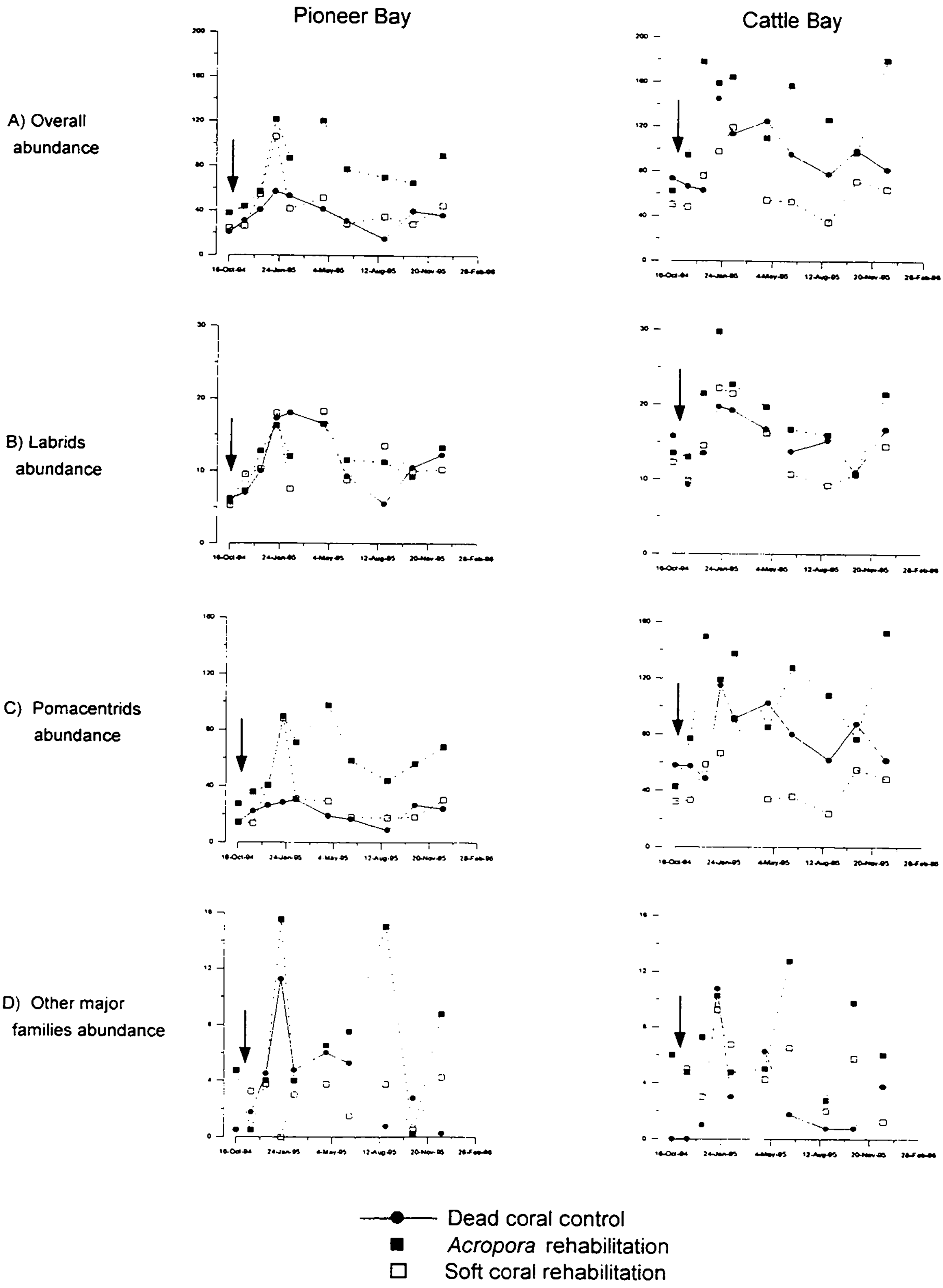


Fig. 7.7. Total abundance (mean no. fish/25 sqm.) of A) Overall fish, B) Labrid, C) Pomacentrid and D) Other major family assemblages during habitat rehabilitation experiments at two localities on Orpheus Island. Arrows indicate time of manipulation.

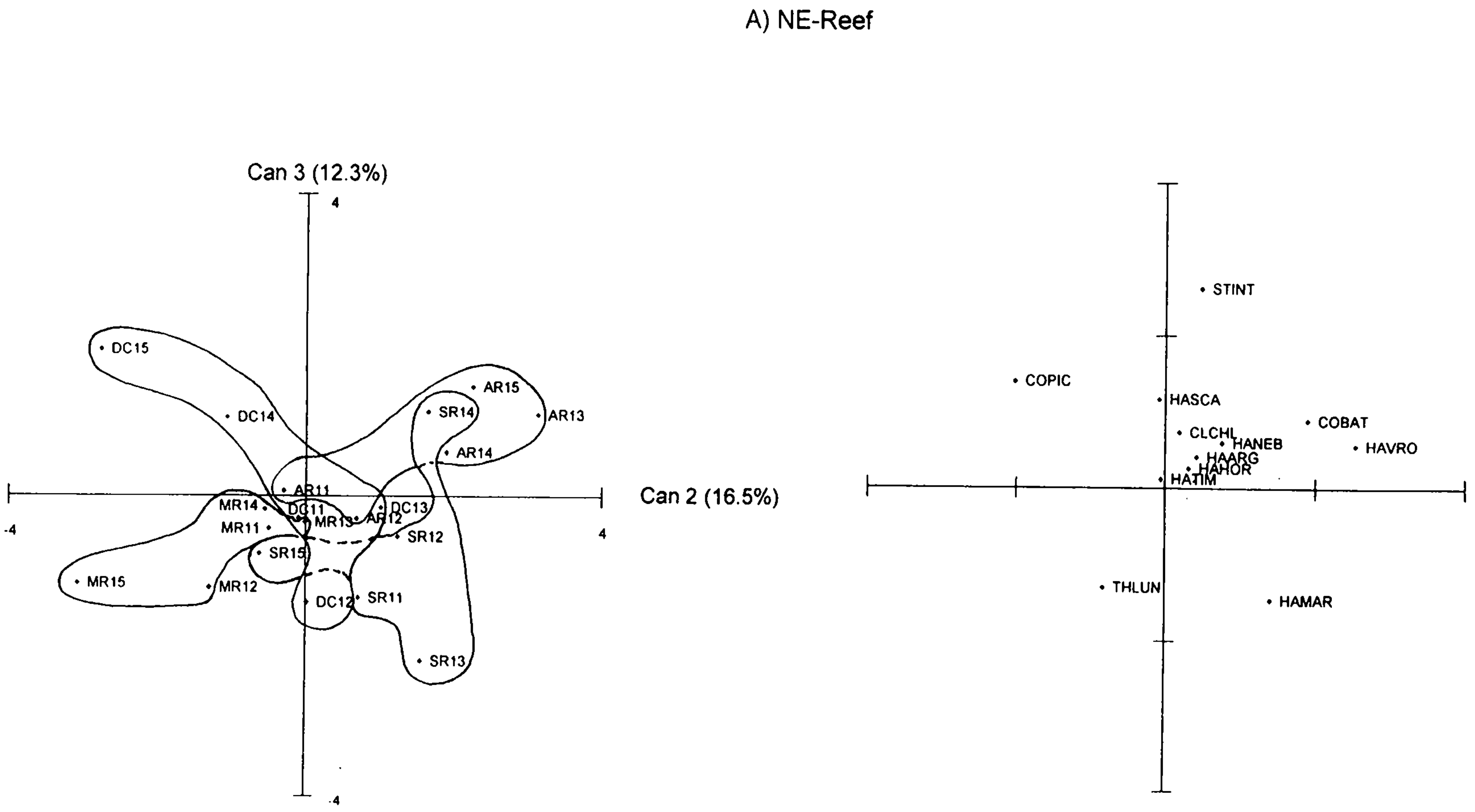


Fig. 7.8A. Ordination plots from CDA showing labrid assemblage responses to habitat manipulation from habitat rehabilitation experiment at NE-Reef on Maiton Island. Treatment code name indicate three factors: first two letters represent treatment - DC = Dead coral control, AR = Acropora rehabilitation, MR = Massive coral rehabilitation, and SR = Soft coral rehabilitaion; third number represents localities - 1 = NE-Reef and 2 = SE-Reef; last number represents sampling time. Fish code name are described in Appendix 7.I.

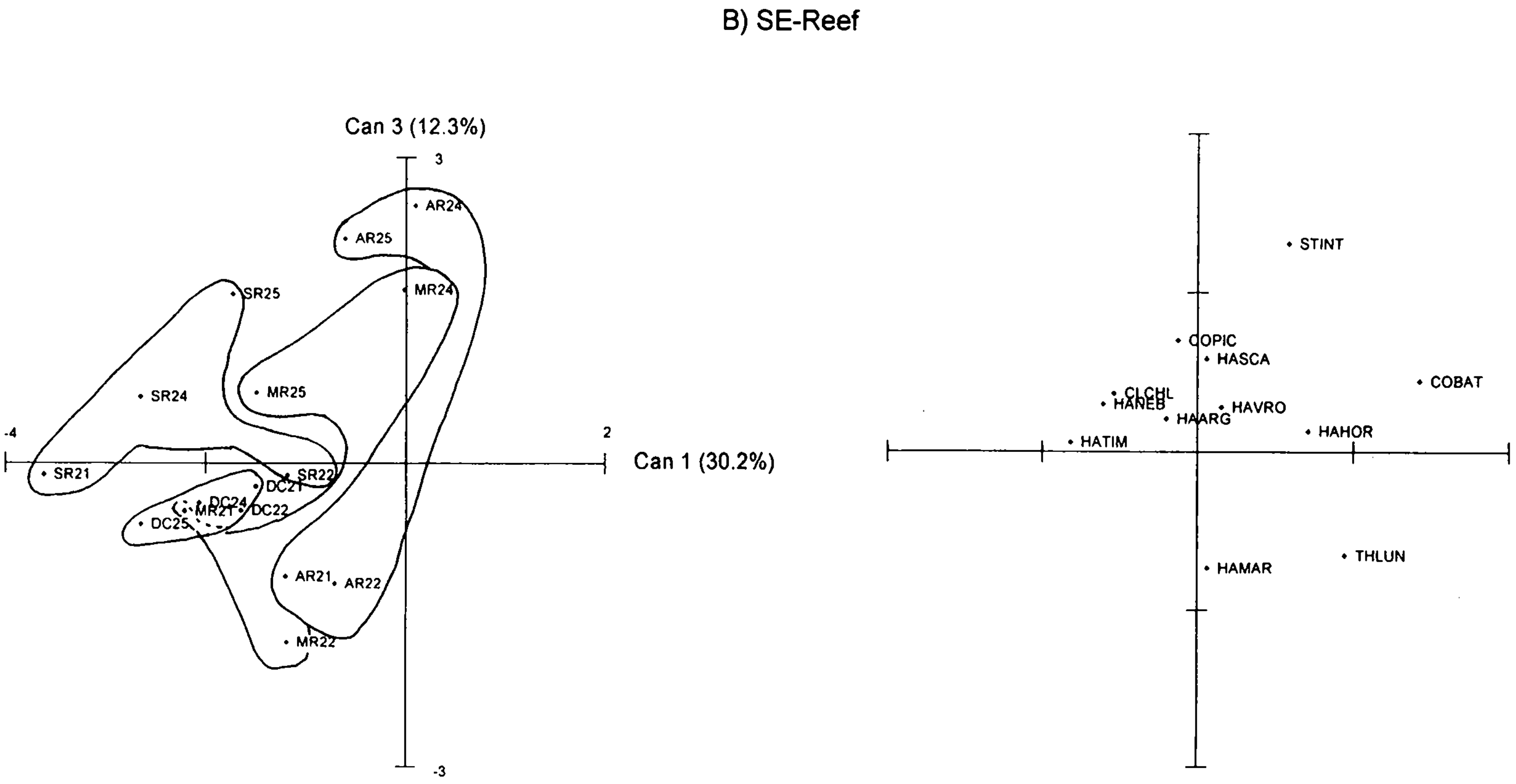


Fig. 7.8B. Ordination plots from CDA showing labrid assemblage responses to habitat manipulation from habitat rehabilitation experiment at SE-Reef on Maiton Island. Treatment code name indicate three factors: first two letters represent treatment - DC = Dead coral control, AR = Acropora rehabilitation, MR = Massive coral rehabilitation, and SR = Soft coral rehabilitaion; third number represents localities - 1 = NE-Reef and 2 = SE-Reef; last number represents sampling time. Fish code name are described in Appendix 7.I.

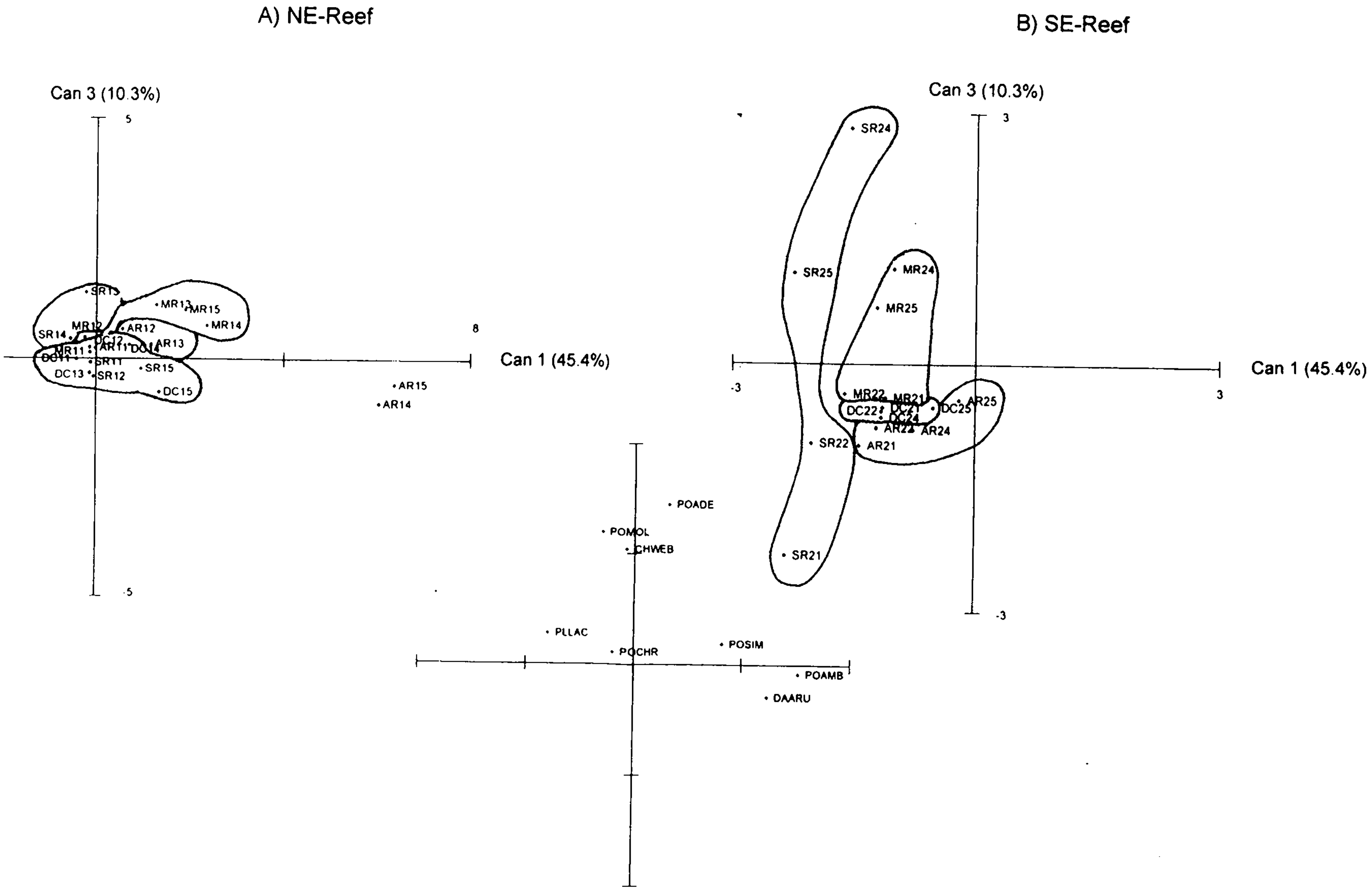


Fig. 7.9. Ordination plots from CDA showing pomacentrid assemblage responses to habitat manipulation from habitat rehabilitation experiment at two localities on Maiton Island. Code names same as Fig. 7.8.

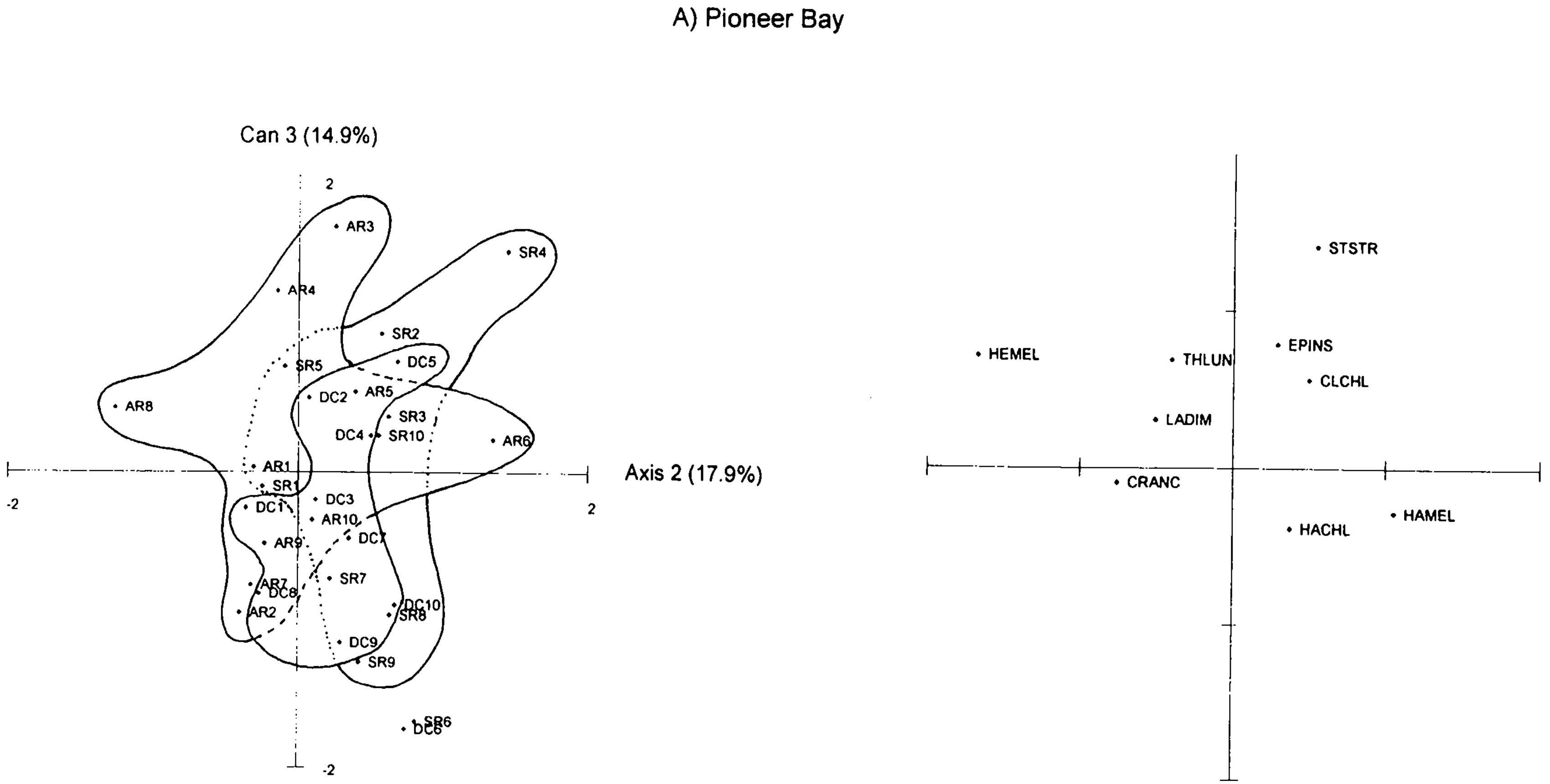


Fig. 7.10A. Ordination plots from CDA showing labrid assemblage responses to habitat manipulation from habitat rehabilitation experiment at Pioneer Bay on Orpheus Island. Treatment code names as in Fig. 7.8 excepted number represents localities, 1 = Pioneer Bay and 2 = Cattle Bay.

B) Cattle Bay

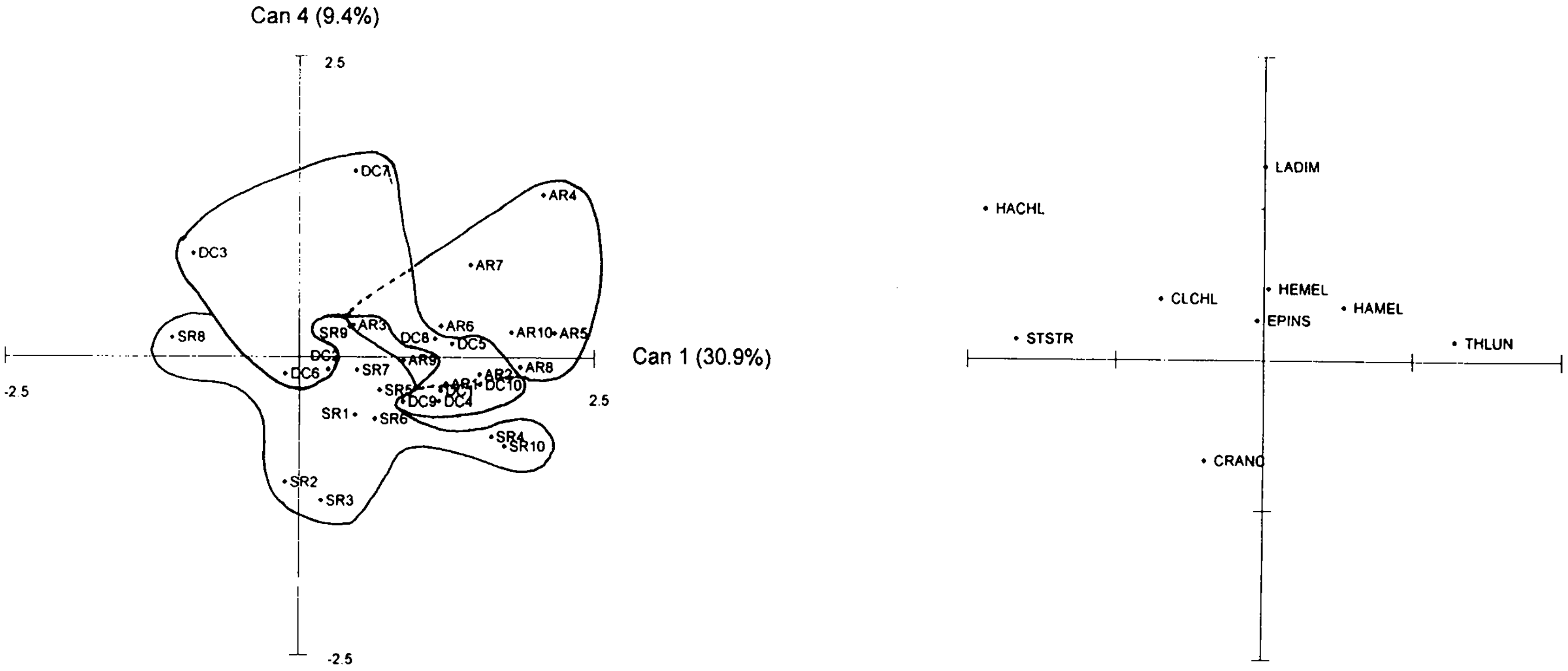


Fig. 7.10B. Ordination plots from CDA showing labrid assemblage responses to habitat manipulation from habitat rehabilitation experiment at Cattle Bay on Orpheus Island. Treatment code names as in Fig. 7.8 excepted number represents localities, 1 = Pioneer Bay and 2 = Cattle Bay.

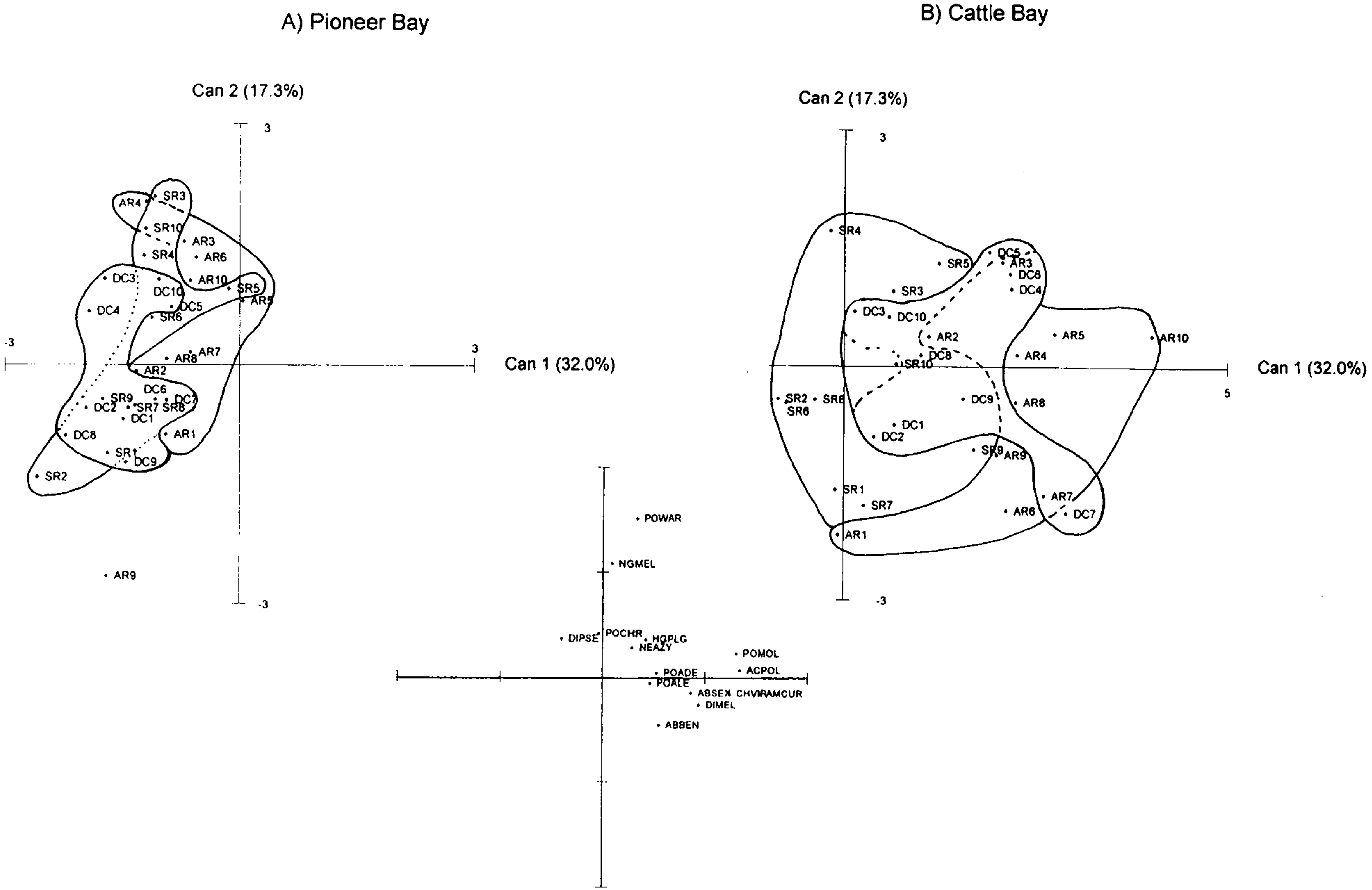


Fig. 7.11. Ordination plots from CDA showing pomacentrid assemblage responses to habitat manipualtion from habitat rehabilitation experiment at two localities on Orpheus Island. Code names as in Fig. 7.10.

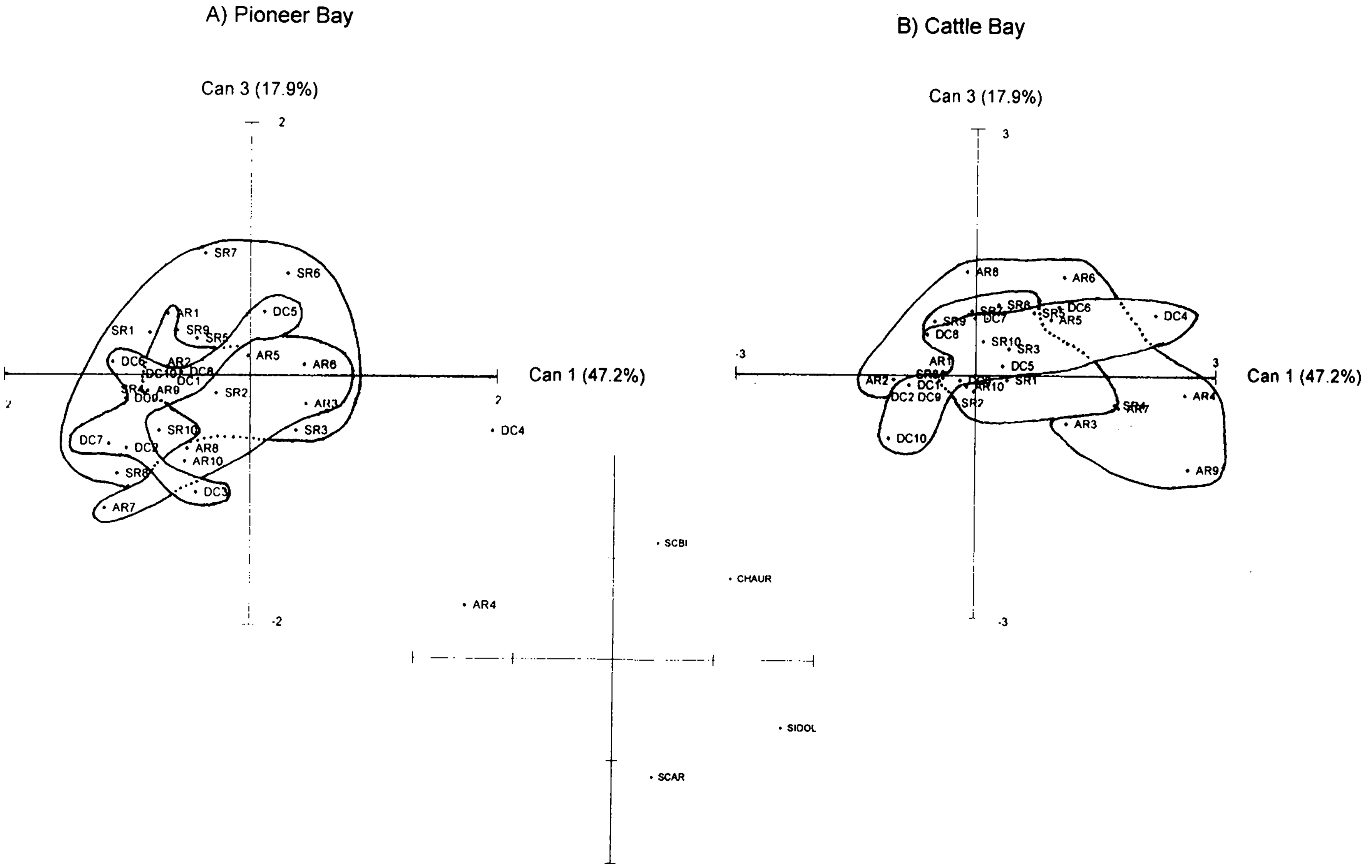


Fig. 7.12. Ordination plots from CDA showing assemblage of other major families responses to habitat manipulation from habitat rehabilitation experiment at two localities on Orpheus Island. Code names as in Fig. 7.10.

# CHAPTER 8

# MULTI-LEVEL INDICES ASSESSING THE STATUS OF CORAL-REEF ASSEMBLAGES: A TOOL FOR CORAL REEF MANAGEMENT

## 8.1 SUMMARY

Decisions for management of coral reef systems on a large spatial scale are often made on the basis of limited biological data, often simply coral cover. However, the use of living coral alone as an index is inadequate because 1) coral represents only part of the ecosystem, 2) the substratum suitable for coral development varies from site to site, and 3) low coral cover may be normal for a particular area subject to high natural disturbance. A new assessment procedure is proposed based on the whole benthic reef assemblage. This procedure involves two major aspects; measures which are used as indices, and a decision rule which is used to transform index values to management information. The measures at assemblage level are based on indices corresponding to suitable habitat and cover data. The four indices are: 1) the "Development Index" (DI) - indicating the level of reef development; 2) the "Condition Index" (CI) - indicating the level of living coral (adjusted to reef development; 3) the "Algal Index" (AI) - indicating the cover of algae on dead substrata and 4) the "Other fauna index" (OI) - indicating the cover of other fauna (e.g. soft coral, sponges and zooanthids) on suitable substrata. The management decision rule proposed is based on five quality classes which is symmetrically classified from a percentage scale: very poor (>0 - 20%), poor (>20% - 40%), moderate (>40 - 60%), good (>60 - 80%), and very good (>80 - 100%). Only one secondary level index is developed by integrating Development Index and Condition Index. The integration employs a multi-dimensional ranking method which also provides for possible development of indices at higher levels, e.g., ecosystem to managerial levels. The application of this new procedure was carried out in three geographical regions: the east coast of the Gulf of Thailand, the waters surrounding Phuket Island and the central Great Barrier Reef. The results of this new assessment procedure provide a new perspective and more practical interpretation than previous assessment from percentage cover data alone. The final result is a quality rank for

each study site which can be used to set up a priority list for management e.g. as an aid in reef zoning.

## 8.2 INTRODUCTION

Assessing the current status of coral reef resources is an obvious preliminary step towards successful management. Most attention from scientists is concentrated on assessing long-term trends from monitoring programs (Brown & Howard, 1985; Brown *et al.*, 1990; Moran *et al.*, 1991; Wilkinson *et al.*, 1993; Chansang & Phongsuwan, 1993) and relies on sophisticated statistical analyses (Warwick *et al.*, 1990; Clarke *et al.*, 1993). Unfortunately, coral reefs in most developing countries are under stress, but long-term data are likely to be limited. Even if monitoring programs are introduced, it may be too late to introduce management plans to halt further damage. Existing data must, therefore, be used as indicators of the status of resource/environment quality, while future monitoring will provide time series data for trend assessment which can be used for evaluating the effectiveness of management measures (Harding, 1992).

Assessment of the health of coral reefs is usually made utilizing either a direct approach - usually a simplified measure of cover for both living and dead corals (Brown & Howard, 1985; Wilkinson *et al.*, 1993), or an indirect approach, e.g., using butterflyfishes as an indicator (Reese, 1981). For entire coral reef systems, a single measure or indicator is inadequate, especially on a large spatial scale. The roles played by other components of a benthic assemblage have frequently been overlooked. Three major problems associated with the application of the percentage coral cover only are: 1) consideration is given to part of the ecosystem, while other types of organisms/substrates are ignored, 2) the same level of reef development is assumed whereas the area suitable for coral growth may vary from site to site, and 3) low coral cover is assumed to be not necessarily unhealthy but in fact this depends on what else occupies the space. It is, therefore, necessary to use a wider number of measures within the ecosystem.

To illustrate the problem with coral cover data alone, consider the data set of the area cover of five major benthic components (Table 8.1). Consideration of living coral alone will give the impression that all 12 reefs are of the same status. When dead coral is taken into account, four reef groups can be distinguished depended on the degree of dead coral present; A - reef 1, B - reef 10 and 11, and 12, C - reef 4, 5, and 6, and D - reef 2, 3, 7, 8, 9. As further biological criteria are taken into account, significant different among the 12 reefs are clearly appearance. The different perspectives on the measures used provide different result of reef status and could stimulate certainly different management decisions. Therefore, an assessment of the status of coral-reef assemblages should provide a clear and understandable result or at least an indication of stress.

The semi-quantitative expression of biological information is another important tool for management as ultimately managers need to rank sites in terms of overall "quality". However, the decision rule used for assessing the quality of resources from quantitative data vary even among studies conducted in the same region or same country (Table 8.2). This is important because different rule will give different results, resulting in different management decisions. This becomes clear when applying the rules in Table 8.2 to the data set in Table 8.1. Considering living coral alone, for example, the decision for all reefs can be: 1) fall at the border line of moderate and good health (Chua & Chou, 1992; Jaker & Looman, 1995); 2) good health (Wilkinson *et al.*, 1993); and 3) abundance of living coral or frequently present (Toivonen & Huttunen, 1995). When the decision rule of Phongsuwan & Chansang (1992) is applied on living coral and dead coral plus dead coral with algae, two reef groups: 1) moderate [reef 1, 2, 4, 5, 6, 7, 8, and 10]; and 2) good [reef 3, 9, 11, and 12] can be recognized. It is, therefore, necessary to set up a new standard for coral-reef assemblage status assessment in term of biological measures and management decision rule.

A more integrated approach, based on the development of an index of resource/environment status, for management, has been implemented in other systems. For example: 1) the development of the Index of Biotic Integrity for water resources using fish (Karr, 1981, 1991; Harris, 1995) and macroinvertebrates (Armitage *et al.*, 1983; Chessman, 1995), and 2) the development of an Integrated Environmental Index for land-use zoning (Sol, *et al.*, 1995). Indices of coral reef health can also be developed by using this approach. However, in most circumstances only the benthic reef-assemblage data are available.

This paper proposes an alternative way of assessing the quality of coral-reef assemblages. All of the major life-form categories are considered in an assessment of the health of coral-reef assemblages. An integration method is also proposed to corporate multiple measures. A decision rule to translate the scientific data into management information was also proposed. Development of the indices and decision rule, their meaning, rationale, application for management are discussed.

## 8.3 MATERIALS AND METHODS

### 8.3.1 STUDY AREAS

The status of coral-reef assemblages was evaluated in three different geographical regions (Fig. 8.1): the Eastern side of the Gulf of Thailand - EGT (South China Sea - Thailand), the waters surrounding Phuket Island - Phuket (Andaman Sea - Thailand) and the central Great Barrier Reef - central GBR (Southwest Pacific Ocean - Australia). In EGT, 24 localities were studied during 1990-1991. These were at the Lan (9), Phai (4), Samaesan (4) and Samet Islands (7). At Phuket, nine localities were studied during March 1994. On central GBR, 14 localities were studied from November 1993 to January 1994. These were at Dunk Island (1), Orpheus Island (9) and Magnetic Island (4). At Phuket and central GBR, two habitats (middle reef slope and reef edge) were studied for each locality except for Dunk Island where only the reef slope was studied. The middle reef slope was usually at a depth of 6-8m below mean sea level (MSL) whereas the reef edge was 3 - 4 m below MSL. On EGT only one habitat, upper reef slope, was studied, usually at a depth of 3 - 4 m below MSL. The difference in depth of similar habitat between EGT and Phuket & central GBR is due to a lower degree of reef development in the Gulf of Thailand (Sakai *et al.*, 1986).

The term "study site" will be used for each habitat at each locality. All of the study localities can be considered as near-shore environments which usually have a high level of influence from land and human activities on reef status.

### 8.3.2 SAMPLING TECHNIQUES

Coral-reef assemblage data were collected using the line intercept transect technique (Loya, 1978; Dartnall & Jones, 1986). However, the benthic life-form or physiognomic structural data based on 21 life-form categories were used instead of the taxonomic levels of benthic organisms. The classification of all 21 life-form categories into higher component summarizes the role of each categories on benthic coral reef-assemblage (Fig. 8.2). Along EGT, data were collected using a single 100 m line transect. At Phuket and central GBR, data were collected using 30 m line transects with six replicates for each study site. Transect lines were laid parallel to the shore while the distance between replicate transects being at least 30 m.

The percentage area cover of five major categories of benthic life-forms: living coral, dead coral, algae, other fauna and abiotic substrata was calculated. This mean percentage area cover was used to calculate the value of each index (formulae described below) of the coral assemblages.

### 8.3.3 FORMULATION OF PRIMARY LEVEL INDICES

The formulation of four indices: 1) the "Development Index" - DI; the "Condition Index" - CI; the "Algal index: - AI; and the "Other fauna index" - OI, are derived from the proportions of key components. These key components may be represented by one or more of the five major benthic categories. However, the index value calculated from the proportion of two key components provides an impractical scale which is difficult to understand and interpret. It is necessary to rescale this to more suitable and practical scale. Therefore, the formula of each index is based on a unit scale (percentage equivalent) expression. By using the same principle, the application of these three indices can be conducted with the same decision rule. These four indices and their formulas are as follows:

#### 8.3.3.1 Development Index (DI)

The DI is used to indicate the degree of coral-reef assemblage development in terms of the area cover of the coral component (CC) or the inverse of non-coral component (NC). For the benthic lifeform area cover data, coral component is represented by live coral, dead coral, algae and other fauna. However, algae and other fauna are included in coral component only when they colonize on a coral component (alive or dead). If they colonized non-coral substrata, they are excluded from the coral component and included in non-coral component. It is very important to note this phenomenon and then use the appropriate formula. Therefore, the DI formula can be expressed in several forms by the effect of substrata origin. In this study, the nature of the substrate beneath "algae" and "other fauna" was not recorded. For the purposes of developing indices, however, algae and other fauna were assumed colonize on "dead coral". The DI formula is as follows.

$$\mathbf{DI} = \mathbf{CC / (CC + NC)}$$

or

$$\mathbf{DI} = \mathbf{(LC + ADC + Algae + OT) / (LC + ADC + Algae + OT + Abiotic)}$$

or

$$\mathbf{DI} = \mathbf{1 - Abiotic}$$

where,

LC = Relative area cover (unit scale) of living coral
ADC = Relative area cover of all dead coral
Algae = Relative area cover of algae
OT = Relative area cover of other fauna
Abiotic = Relative area cover of abiotic non-corals e.g. sand, gravel and rock

#### 8.3.3.2 Condition Index (CI)

The CI is used to assess the condition of coral-reef assemblages in terms of living coral cover but adjusted to the degree of reef development. To assess the condition of coral reefs, only coral component should be considered. Non-coral component should not be included in the formula for condition assessment because abiotic components do not have a direct meaning in terms of the reef condition as they do for reef development. For example, take two study reefs; the first reef has a live coral cover of 50% and a dead coral cover of 50%, while the second reef has a live coral cover of 50% and a sand cover of 50%. Both reefs have the same live coral cover, but their status is different when dead coral and sand are taken into account. The dead coral present on the first reef indicates that there is some disturbance (natural or anthropogenic over short or long time periods) causing coral death. The absence of dead coral on the second reef indicates that there is no observable disturbance on this reef. The absence of sand (or any non-coral component) on the first reef indicates that corals can develop over all of the reefal area. In contrast, the

50% sand cover indicates that corals cannot cover all of the area and therefore development of this assemblage is not as high within this reef. The formulation of CI was, therefore, only considered from coral component by using the proportion between live coral related component (LRC) to dead coral related component (DRC). The formula of CI expressed as a unit value is as follows.

$$\mathbf{CI} = \mathbf{LRC / (LRC + DRC)}$$

or

$$\mathbf{CI} = \mathbf{LRC / (LRC + ADC + Algae + OT)}$$

(using the same abbreviations as for DI)

#### 8.3.3.3 Algal index (AI)

The AI is used to indicate the potential of algae for occupying available space which is usually on dead coral. The formulation of this index, therefore, is based on the dead coral related component only. In this study, however, dead coral related component are comprised of three major categories, dead coral, algae, and other fauna. The formula of the index is as follows.

$$\mathbf{AI} = \mathbf{Algae / (Algae + ADC + OT)}$$

(using the same abbreviation as for DI)

#### 8.3.3.4 Other fauna index (OI)

The OI is used to indicate the potential of other benthic fauna for occupying available space (dead coral). Therefore, the formulation is based on the same principle as AI. The formula of the index is as follows.

$$\mathbf{OI} = \mathbf{OT / (OT + ADC + Algae)}$$

(using the same abbreviation as for DI)

The application of these indices for comparison can be made only within the same habitat because different habitats have different degrees of environmental stress which cause different reef development and conditions. Thus, the results of reef edge and upper reef slope were considered separately.

### 8.3.4 MANAGEMENT DECISION RULE

The primary goal of these indices is to display the status of coral-reef assemblages for coral reef resource management. Therefore, it is necessary to transform the quantitative data into qualitative information. The rule used to justify the quantitative data is a subjective aspect which vary between workers (Table 8.2). The rule used by one worker may not be accepted by others. For general acceptance, therefore, the initiation of rule must be constructed using a more systematic approach to minimize the bias within the rule itself. Scales used for rule classification expressed in four corresponding forms; percentage, ratio, proportion and the log-transformation of proportion are shown in Table 8.3.

Basically, this study uses a "balanced quality" and "symmetrical quantity" classification based on percentage value . Balanced classification can be achieved by assigning equal positive and negative quality classes with intermediate quality in between, e.g. the three classes rule; poor, moderate, and good, is balanced while the rule with four classes; poor, moderate, good and very good is not balanced because bias at superlative side (very good). Symmetrical classification can be achieved by dividing each quality class with the same range of quantity values, e.g. five classes rule will have the value of 20% for each class when use percentage scale. Thus, the quality of each index was classified into five classes; very good (1), good (2), fair (3), poor (4) and very poor (5) which equate to the quantity value range of each class being 20%. A finer level of classification can also be made if required by dividing each class into two subclasses, positive and negative, e.g. good+ and good-. Zero value is not included in the rule because it means absence which should not be evaluated with several degrees of presence.

### 8.3.5 DEVELOPMENT OF SECONDARY LEVEL INDICES

Secondary level indices are actually a new quality rank which is integrated from two or more primary indices. Only one index is proposed by integrating Development Index and Condition Index to give the Coral Assemblage Status Index (CAS Index). The minor components like algae and other fauna are not included in CAS Index because of their uncertain relationship with the health of coral reefs.

The procedure used to develop the secondary index level is based on the "Multi-dimensional Ranking" method which ranks from more than two measures (variables). The graphical procedure of multi-dimensional ranking with three different scales is demonstrated in Fig. 8.3. The original data (Fig. 8.3A) can be used when the exact position is needed but it is not practical especially for large spatial scales. The semi-quantitative transformed data or quality rank (Fig. 8.3B) is likely to be more practical than the original data. This ranking procedure is that the number of classes within a new integrated decision rule increase depending on the number of measures and the number of classes within decision rule of each measure. This causes each rank of the new classes to lose its logical meaning and the new decision rule may loose its balance. Fig. 8.3C demonstrates the integration of index X (9 classes) and index Y (5 classes) providing a new index with 13 classes. It is necessary to provide a number of the final class and define the lowest and highest class clearly as a reference scale.

This graphical method can be achieved effectively only for two measures. When there are more than two measures, a number of measures minus one (n-1) steps are needed for graphical assessment which makes the process much more complicated. For example, if three measures are required to be integrated, the first two measures must be performed as in Fig. 8.3A or Fig. 8.3B. The result obtained from this step will be considered with the third measures as shown in Fig. 8.3C. An alternative way is by calculation which is much more simple. The number of classes of integrated decision rule can be calculated from the following formula.

$$\textbf{No. of integrated classes} = (x_1+x_2+\ldots+x_n) - (n-1)$$

where, $x_n$ = number of classes within decision rule of measure/index n

$n$ = number of measures/indices used

The final integrated rank can be calculated from the following formula.

$$\textbf{Final integrated rank} = (r_1+r_2+\ldots+r_n) - (n-1)$$

where, $r_n$ = rank of measure n

$n$ = number of measures used

## 8.4 RESULTS

### 8.4.1 BENTHIC AREA COVER AND PRIMARY LEVEL INDICES

The detailed analyses for all three geographical regions in term of the coral-reef assemblage structure (habitat structure) have been described elsewhere (CREST, 1989; Chapter 4 and 5). The following results were, therefore, considered from the major benthic categories and the indices relevant to them by focusing on the application for management.

The regions differed substantially in terms of the degree of reef development (Fig. 8.4). Along the East of the Gulf of Thailand (EGT), Development Index (DI) varied between 0.4 and 0.1. In contrast, most of the study sites at Phuket and central Great Barrier Reef (central GBR) had Development Index between 0.8 to 1.0 except three sites; O1S, D1S and P8E where the Development Index fell between 0.6 to 0.8. These results indicate that reef development at the EGT varies from moderate to very good development while at Phuket and the central GBR almost all of the study sites had very good development.

The mean live coral cover and Condition Index (CI) differed among the geographical areas (Fig 8.5). Along the EGT, the correlation between live coral cover and Condition Index was significant ($r = 0.7244$, $P < 0.05$) but it was not as highly correlated as at Phuket and central GBR. Out of 24 sites, 11 obtained a different semi-quantitative assessment. Almost all the values of live coral cover and Condition Index from Phuket and central GBR were close together except at the reef slope of Dunk Island. The correlation between live coral cover and Condition Index in these two regions, for both at the reef slope and the reef edge, were very high (reef slope $r = 0.9864$, $P < 0.001$ and reef edge $r = 0.9982$, $P < 0.0001$). The semi-quantitative assessment of live coral cover and Condition Index were, therefore, mostly the same except for P4S and P4E at Phuket and O4S, O7S, M4S and D1S at central GBR.

Algae were common only at Magnetic Island while they were uncommon at other study areas (Fig. 8.6). At Magnetic Island, all results of semi-quantitative assessment of algae cover and Alagal Index were different. Three sites were found to have low algal cover; M1S, M4S and M3E, while the rest were found to have moderate cover. For Algal Index on the other hand, two sites were good while the rest were very good. This result indicated the high to very high potential for algae to occupy available space while actual cover was in fact low to moderate.

Other fauna was found mostly at Orpheus Islands, especially on the reef edge and on some sites along the EGT (Fig. 8.7). The existence of other fauna on most sites at Orpheus Islands was low to moderate, but the Other fauna Index indicated various degrees to cover available space; the potential was very high at O3E, high at O3S, O1E and O4E, and moderate at O1S, O2S, O6E, O8E and O9E. Along the EGT, the Other fauna Index indicated moderate potential at G8S and high at G1S and G15S but the actual existence of other fauna at most sites was very low.

A series of scatter plots between Condition Index and other three indices (Development Index, Algal Index and Other fauna Index) provide a practical interpretation on the status of each site (Fig. 8.8). The results indicated that these three regions have some different features. The reefs of Phuket do not have a prominent potential occupation by algae and other fauna, while those of the EGT have for other fauna, and those of central GBR have both algae and other fauna. From this series plot, however, the status of each site in each region can be described according to all five major components by sequentially considering from Development Index, Condition Index, Algal Index and Other fauna Index. In the EGT, for example, G9S was covered by reef component only *ca.* 50%, among reef component *ca.* 15% were dead coral (or degraded by any means) which only a small part (*ca.* 21%) were occupied by algae while other fauna were rare or not present. For Phuket, on both reef slope and reef edge of P1 and P2 were all covered by reef component (100%) but *ca.* 80% were dead coral only. For central GBR, M4S were covered by reef component *ca.* 90% , among this component *ca.* 40% were dead coral which mostly occupied by algae (*ca.* 80%) and rarely occupied by other fauna (less than 10%).

### 8.4.2 SECONDARY LEVEL INDICES

Plots between DI and CI of study sites which illustrates the actual position of each study site, when ranking it is actually CAS Index. The rank classification of study site can be created based on CAS Index (Table 8.4). This result shows that reefs with equal or higher living coral cover do not necessarily have the same or better ranking than a site with lower living coral cover e.g. between G2S & G17S, G11S &

G1S, O4S & O7S, and O2S &O8E. The status of the coral assemblages of the same habitat between geographical regions can also be considered.

## 8.5 DISCUSSION

### 8.5.1 INDICES RATIONALE

This paper proposes an alternative way of assessing the quality of coral-reef assemblages and sets out standard criteria to translate the scientific (quantitative) data into management (qualitative) information. Four indices: Development Index, Condition Index, Algal index and Other fauna index, incorporated with a series plot, provide a new perspective to look at the assessment of the status of a coral-reef assemblages. These indices are based on the concept that cover data must be adjusted on the basis of the area of suitable substrata available to different organisms. Therefore, the values of these indices indicate the relative potential of each organism/group to occupy available space.

These indices were designed as a tool for management to aid decision making on reef status in term of semi-quantitative and qualitative. The calculation of the indices could be done for each observation and then averaged as a mean for each site. But the problem is this mean index would come with variation (SD or SE) which causes the problem when transformed mean index into quality class. This variation will cause the quality class to be unstable, especially at the upper and lower limits. This causes difficulty for management decisions.

The development of most of the study sites surveyed was very good, the values between live coral cover and Condition Index are close except along the EGT (Fig. 8.5). However, Condition Index provided a higher value than area cover data and in some cases provided a different qualitative assessment result. For algal cover - Algal Index and other fauna cover - Other fauna Index, a difference between area cover and index existed for some areas, e.g., algae at Magnetic Island (Fig. 8.6) and other fauna at Orpheus Islands (Fig. 8.7). The difference in these results comes from the nature of the substrata variables which change from site to site, especially in the Gulf of Thailand which receives the effect of water run-off from four main rivers. Abiotic non-coral components (illustrating the degree of reef development) are the most important variables for the application of Development Index and Condition Index. If there are no or very low levels of abiotic non-coral component, Condition Index is identical to live coral cover. The absence of an abiotic non-coral component does not affect Algal Index or Other fauna Index, but the abundance of algae and other fauna do. Therefore, Algal Index and Other fauna Index are more useful when algae and other fauna are present in some degree.

Development Index and Condition Index are the most useful indicators of low reef development in situation such as nutrient rich-near-shore coral communities (Sakai *et al.*, 1987) or subtropical coral communities (Harriott *et al.*, 1994) where significant limestone reef platforms are not formed (Veron & Done, 1979; Sakai *et al.*, 1987). The sampling technique is also important for the application of these two indices and should be based on the stratification concepts, such as the belt transects (Loya, 1978) rather than the selection of the highest coral cover area (Harriott *et al.*, 1994). This is important because different habitats tend to have different levels of development and experience different stresses. The selection of the highest coral cover, by definition results in selection of "unrepresentative" habitat.

Condition Index show clearly application on a wide range of condition. For Development Index, Algal Index and Other fauna Index, their validity appear to be unclear. Their application must be, therefore, done with caution. The problem of Algal Index and Other fauna Index was depended on their uncertain relationships of algae and other fauna with the stable state of the reef. Their application may be used as a supporting indicator. For Development Index, its application depend on the nature of the reef and sampling error. It always appeared on some reef habitat (zone), especially reef flat, of the true reef (limestone platform) that some spot is occupied by sand (transported by wave action). Some sampling techniques are allowed to record WATER for the interval in which the benthic substrata are deep below the transect line (over 50 cm) (English *et al.*, 1994) and also record for missing data. In both cases, Development Index is not valid and must not be interpreted as index for reef development but should be regarded as bias. These problems might showed that the application of Condition Index is essential and more appropriate than that of living coral.

### 8.5.2 DECISION RULE RATIONALE

The decision rule used to assess the quality of reef resources is important for management. For environmental impact assessment, a statistical decision is a common practice (Mapstone, 1995). To assess the health of the existing resources or environment, a descriptive decision is likely to be the most suitable way because the major objective is not to detect any changes but to show recent status. Management decisions should be made without variation while scientific (ecological) data are always variable. If the original data set is used to assess resource quality, it may not be easy to make a decision. Therefore, it is necessary to transform scientific data into management information by setting up a rule to do so. Scientists working on each particular resource should ideally be the ones who set up that rule rather than resources managers alone (Underwood, 1995).

The decision rule used in this study was based on a balanced and symmetric classification to eliminate bias within the rule. Five quality classes were assigned, based on logical meaning with reference numeric values (for further application). The advantage of this classification is that it is simple and easy to understood. It is necessary to remember that management processes involve not only scientists and resource managers but the public as well. It is, therefore, necessary to describe the status of resources in the simplest form possible but not over simplified. The application of the indices can solve this problem because each index is a single value but calculated from a set of relevant variables to it.

One question that needs to be considered is whether zero values should be included or not. The previous work usually included zero among the criteria (Karr, 1991; Sol *et al.*, 1995). However, the zero means absent which can not be weighted in the same way as presence to varying degrees. This means the zero should be used in the evaluation only in a simple presence-absence scale which is important for the monitoring program. An example of another approach is the Biological Monitoring Working Party scoring system (Armitage *et al.*, 1983) that gives a different score (weight) for the different indicator groups present, without respect to abundance, but gives a zero score when absent.

### 8.5.3 SECONDARY LEVEL INDICES

The application of these indices and decision rule for management can be done directly from a series plot of three index pairs; CI-DI, CI-AI, and CI-OI. The description of each site can be described easily from this plot. Condition Index and Development Index should be used as major pair (forming Coral Assemblage Status Index - CAS Index). While Algal Index and Other fauna Index should be used as supporting indicator. Thus, the status of the reef over a spatial scale and also a temporal scale can be ranked and classified which is believed to be an essential part for management.

The secondary and higher level indices must be developed from the measures that have the same quality direction on reef status. The CAS Index, for example, are developed by integrating Development Index and Condition Index together where both indices have the same directional rank (1 indicates highest while 5 indicates lowest status). The measures involving algae and other fauna are not included in the CAS Index because their relationship with reef status is uncertain. If they have a negative relationship, it is still possible to include them in the CAS Index by redirecting the rank.

For a small spatial scale where reef development is all the same, live coral cover may be used directly instead of CAS Index. For a large or very large spatial scale, CAS Index can be used to provide a broad picture of the health of the coral assemblages. The ranking and multi-dimensional ranking methods are proposed rather than scoring and simple summation (Karr, 1991) or power formula summation (Sol *et al.*, 1995) because ranking and integration methods respect the equal (logical) quality of each index/measure while scoring and summation methods do not.

### 8.5.4 HIGHER LEVEL INDEX

The management of the environment is likely to deal with many issues including ecological process, decision philosophy (Fairweather, 1993), economic analysis of natural resources (Underwood, 1995), social and politics (Mapstone, 1995). The highest decision level, therefore, should be considered with all of these components. A higher index level can be developed as in the model proposed in Fig. 8.9 by using the multi-dimensional ranking method as a tool. These highest decision level comprise of two major components (levels), ecological and managerial, then call

"Man-Eco Decision level". For ecological side, a number of elements can be taken into account such as coral-reef assemblage, reef fish assemblage and environmental status. In each element, indicator or a group of indicators will be used to indicate the status of each element. In this study, therefore, is the development of the only one element (coral-reef assemblage) in the model.

For managerial component, it comprises of two major sub-components; socio-economic status and management policy. Socio-economic status can be fisheries, tourism and culture factors which may vary among areas. All of the measures involved with these managerial factors can be integrated with any level of index from the ecological side which will provide higher index (decision) level.

For example in the Gulf of Thailand, if the tourism industry along the coast line is well established and local people rely on them, it is possible that the policy to manage the reef resources will be pro-tourism rather than fisheries or other activities. Therefore, the reefs may also be ranked based on their potential for tourism (how to get this rank is beyond the scope of this paper). This rank for tourism then can be integrated with the ecological index.

Management decisions can be made at different levels depending on the information to hand. The selection of variables included in an assessment process should not provide the same information. If the same information variables are used, it causes the dilution of the information (assessment) by increasing the ranking order without meaning.

### 8.5.5 LIMITATION

These proposed indices are still an artificial indicator for the status of coral reef ecosystem. A high value does not always mean a "healthy" reef and conversely a low value does not always mean it is "unhealthy". This is because they are based on coral reef benthic assemblage alone. The relationships between coral reef benthic assemblage with other assemblages might also play a key role on the status of the coral reef ecosystem. However, this relationship is in doubt e.g. coral reef benthic with fish assemblages (Sale, 1991b; Roberts *et al.*, 1992; McClanahan, 1994) and might not be linear as many expected (Chapter 4 and 5).

The best evaluation of the health of coral reefs should be based on long term monitoring. For example, the BACIP (Before-After-Control-Impact-Pairs) designs

(Underwood, 1992; Faith *et al.*, 1995) should be applied for monitoring programs if possible. However, to conserve natural resources, management policy should be more assertive/pro-active. A "wait and see" policy, as the application of monitoring programs after significant changes have been detected, is in most cases likely to be too late. The management of coral reefs must be conducted at the beginning of the monitoring program and up dated when most recent data are available. This time series data can then be used for evaluation of management implementation.

### 8.5.6 CONCLUSION

The present problem of coral-reef assemblage status assessment is the way the data are explained. This paper presents both concept and methods of multi-level indices which is believed to be a practical tool for coral-reef management. A primary assessment based on benthic assemblage is reasonable because it is a bio-physical feature that provides living habitat structure. The decision on its status based on living coral alone, however, is not enough because of the problem of substrata variation and also because of the bias from sampling technique. An alternative perspective based on relative proportion of major substrata were proposed in the form of indices as well as the development of the criterion to evaluate them. An integration approach is proposed to develop more comprehensive index and possibility to be applied for geographical information system. The application, thus, allows management of coral reef at large spatial scale.

Table 8.1. Artificial data set of the area cover as a percentage of five major benthic substrates on the coral reef.

| | Reef | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Variable | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 | 11 | 12 |
| Living Coral | 50 | 50 | 50 | 50 | 50 | 50 | 50 | 50 | 50 | 50 | 50 | 50 |
| Dead Coral | 50 | 0 | 0 | 25 | 25 | 25 | 0 | 0 | 0 | 12.5 | 12.5 | 0 |
| Dead Coral with algae | 0 | 50 | 0 | 25 | 0 | 0 | 25 | 25 | 0 | 12.5 | 0 | 12.5 |
| Other fauna | 0 | 0 | 50 | 0 | 25 | 0 | 25 | 0 | 25 | 0 | 12.5 | 12.5 |
| Sand | 0 | 0 | 0 | 0 | 0 | 25 | 0 | 25 | 25 | 25 | 25 | 25 |

Table 8.2. Qualitative and quantitative scales used by ecologists as decision rule to justified the abundance of organism or the quality of environments/resources.

| Measure | Quality degree | Lowest | ← | | | | | Moderate | → | | Highest | Quali-Quanti scale characteristic | References |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Quantity degree | 6 | 5 | 4 | 3 | 2 | 1 | 0 | 1 | 2 | 3 | | |
| % cover, abundance (Braun-Blanque scale) | | | | | | >5 | 5-25 | 25-50 | 50-75 | >75 | | Balanced - Asymmetric | Jaker & Looman (1995) |
| % cover, abundance | | rare | few | many | abundance | 5-12.5 | 12.5-25 | 25-50 | 50-75 | >75 | | Non-balanced - Asymmetric (mixed scale) | Jaker & Looman (1995) after Barkman *et al.* (1964) |
| Code of Barkman et al (1964)* | | r | + | 1 | 2m | 2a | 2b | 3 | 4 | 5 | | | |
| Score of van der Maarel (1979)* | | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | | Non-balanced - Asymmetric | van der Maarel (1979) |
| % cover | | | | | 0 | 1-5 | 6-30 | 31-50 | 51-75 | 76-100 | | ?** - Asymmetric | Dahl (1981) |
| % cover | | | | | | 1-5 | 5-10 | 11-30 | 31-80 | >80 | | ?** - Asymmetric | Done (1982) |
| ratio coral to other substrates (visual estimate) | | | | | | | little | moderate | dense | continuous | | Non-balanced - ?*** | Findley & Findley (1985) |
| Rating score (quantity varies on measures) | | | | | | | 1 | 3 | 5 | | | | |
| e.g. % of top carnivores | | | | | | | <1 | 1-5 | >5 | | | | |
| % of insectivores | | | | | | | <20 | 20-45 | >45 | | | Incomplete - Asymmetric | Karr (1991) |
| % of omnivores | | | | | | | >45 | 45-20 | <20 | | | | |
| Index of Biotic Integrity | | | | | No fish | 12-22 | 28-34 | 40-44 | 48-52 | 58-60 | | Balanced - Asymmetric with absent | Karr (1991) |
| % Living coral | | | | | | | <30 | 30-50 | 50-75 | >75 | | Non-balanced** - Asymmetric | Chua & Chou (1992) |
| % Living coral / % Dead coral | | | | | | | 1:>2 | 1:2 to 2:1 | >2:1 | | | Balanced - Asymmetric | Phongsuwan & Chansang (1992) |
| % Living coral / % Dead coral | | | | | | 1:>3 | 1:>2 | 1:1 | >2:1 | | | Non-balanced - Asymmetric | Phongsuwan *et al.* (1993) |
| % Living coral | | | | | | | >0 | >25 | >50 | >75 | | Non-balanced - Symmetric | Wilkinson *et al.* (1993) |
| Index score | | | | | >100 with sub-index exceed limit | >100 with no sub-index exceed limit | 75-100 | 75-50 | 50-25 | 25-0 | | Non-balanced - Asymmetric | Sol *et al.* (1995) |
| % Aquatic macrophytes | | | | | <1.5 | 1.5-3 | 3-6 | 6-12 | 12-25 | 25-50 | 50-100 | Balanced - Asymmetric | Toivonen & Huttunen (1995) |
| Frequency | | | | | Very rare | Rare | Fairly rare | Occational | Fairly frequent | Frequent | Very frequent | | |
| Abundance | | | | | Very sparse | Sparse | Fairly sparse | Scattered | Fairly abundance | Abundance | Very abundance | | |

* Jaker & Looman (1995) demonstrated that scale rare to abundance of Barkman *et al.* (1964) and scale 1 to 4 of van der Maarel (1979) are not valid.

** the rule was not qualitatively assigned by the author.

*** the rule was not quantitatively assigned by the author

Table 8.3. Semi-quantitative scale for an assessment of index quality in four corresponding forms: percentage, ratio, proportion and log-transformed proportion. Zero value is treated separately because it means absent.

| Quality (rank) | | Percentage | Ratio | Proportion | Log-transformed proportion |
|---|---|---|---|---|---|
| | Formula | (100x)/(x+y) | x:y | x/y | Log (x/y) |
| Very poor (5) | | >0 to 20% | < 1:4 | < 0.250 | < -0.602 |
| Poor (4) | | >20% to 40% | 1:4 to 1:1.5 | 0.251 to 0.667 | -0.602 to -0.176 |
| Moderate (3) | | >40% to 60% | 1:1.5 to 1.5:1 | 0.668 to 1.500 | -0.175 to 0.176 |
| Good (2) | | >60% to 80% | 1.5:1 to 4:1 | 1.501 to 4.000 | 0.177 to 0.602 |
| Very good (1) | | >80 % to 100% | > 4:1 | > 4.001 | > 0.602 |

Table 8.4. Rank Quality of the study sites based on coral assemblage status index (CAS index). Other measures/indices are also presented for comparision. Site is presented as code name when, the first letter indicates region, excepted for the central section of the Great Barrier Reef used area name; D = Dunk Island, O = Orpheus Islands, and M = Magnetic Island, second number indicates locality number as presented in Fig. 1, and third letter indicates habitat; S = Reef Slope, and E = Reef Edge. DI = Development Index, CI = Condition Index, LC = Living Coral Cover, AI = Algae Occupying Index, Algae = Algae Cover, OI = Other Fauna Occupying Index, and OT = Other Fauna Cover.

| | | | | MEASURE | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| REGION | LOCATION | SITE CODE | CAS index | DI | CI | LC | AI | Algae | OI | OT |
| 1) The east of the Gulf of Thailand (G) | Jun-South | G7S | 2 | 1 | 2 | 2 | 0 | 0 | 5 | 5 |
| | Khungbadan-Northeast | G13S | 2 | 1 | 2 | 2 | 0 | 0 | 5 | 5 |
| | Sak-South | G2S | 2 | 1 | 2 | 3 | 0 | 0 | 5 | 5 |
| | Nok-East | G8S | 3 | 3 | 1 | 3 | 0 | 0 | 3 | 5 |
| | Nok-West | G9S | 3 | 3 | 1 | 3 | 0 | 0 | 4 | 5 |
| | Krok-East | G3S | 3 | 2 | 2 | 3 | 0 | 0 | 4 | 5 |
| | Lan-South | G5S | 3 | 2 | 2 | 3 | 0 | 0 | 0 | 0 |
| | Phai-Southeast | G11S | 3 | 2 | 2 | 3 | 0 | 0 | 0 | 0 |
| | Kharm-North | G17S | 3 | 1 | 3 | 3 | 0 | 0 | 5 | 5 |
| | Chun | G20S | 3 | 1 | 3 | 3 | 0 | 0 | 0 | 0 |
| | Samet-Southeast | G22S | 3 | 1 | 3 | 3 | 0 | 0 | 0 | 0 |
| | Kudee-Northwest | G23S | 3 | 1 | 3 | 3 | 0 | 0 | 5 | 5 |
| | Lan-Southwest | G6S | 3 | 1 | 3 | 3 | 0 | 0 | 5 | 5 |
| | Samet-Southwest | G19S | 3 | 1 | 3 | 3 | 0 | 0 | 5 | 5 |
| | Raet-South | G15S | 3 | 1 | 3 | 3 | 0 | 0 | 2 | 4 |
| | Sak-North | G1S | 4 | 3 | 2 | 3 | 0 | 0 | 2 | 5 |
| | Hin Yuan | G21S | 4 | 3 | 2 | 4 | 0 | 0 | 4 | 5 |
| | Raet-East | G16S | 4 | 2 | 3 | 4 | 0 | 0 | 5 | 5 |
| | Krok-West | G4S | 4 | 2 | 3 | 4 | 0 | 0 | 5 | 5 |
| | Plateen-East | G24S | 4 | 1 | 4 | 4 | 0 | 0 | 0 | 0 |
| | Khungbadan-East | G12S | 4 | 1 | 4 | 4 | 0 | 0 | 4 | 5 |
| | Samet-Northwest | G18S | 4 | 1 | 4 | 4 | 0 | 0 | 0 | 0 |
| | Phai-North | G10S | 5 | 2 | 4 | 4 | 0 | 0 | 5 | 5 |
| | Samaesan-Northwest | G14S | 5 | 1 | 5 | 5 | 0 | 0 | 4 | 4 |
| 2) Phuket (P) | Lon - Southeast | P5E | 1 | 1 | 1 | 1 | 0 | 0 | 0 | 0 |
| | Hi - North | P6E | 2 | 1 | 2 | 2 | 5 | 5 | 0 | 0 |
| | Aoe - North | P7E | 2 | 1 | 2 | 2 | 0 | 0 | 0 | 0 |
| | Lon - Southeast | P5S | 2 | 1 | 2 | 2 | 0 | 0 | 0 | 0 |
| | Aoe - North | P7S | 2 | 1 | 2 | 2 | 0 | 0 | 5 | 5 |
| | Hi - North | P6S | 2 | 1 | 2 | 2 | 5 | 5 | 0 | 0 |
| | Kata | P4S | 2 | 1 | 2 | 3 | 5 | 5 | 5 | 5 |
| | Patong - South | P3S | 3 | 1 | 3 | 3 | 0 | 0 | 5 | 5 |
| | Kata | P4E | 3 | 1 | 3 | 4 | 5 | 5 | 5 | 5 |
| | Patong - South | P3E | 4 | 1 | 4 | 4 | 0 | 0 | 5 | 5 |
| | Kamala | P2E | 4 | 1 | 4 | 4 | 5 | 5 | 0 | 0 |
| | Kamala | P2S | 4 | 1 | 4 | 4 | 0 | 0 | 0 | 0 |
| | Niyang - North | P1S | 4 | 1 | 4 | 4 | 5 | 5 | 5 | 5 |
| | Maiton - East | P8S | 4 | 1 | 4 | 4 | 0 | 0 | 5 | 5 |
| | Niyang - North | P1E | 5 | 1 | 5 | 5 | 4 | 4 | 5 | 5 |
| | Maiton - East | P8E | 6 | 2 | 5 | 5 | 0 | 0 | 5 | 5 |
| 3) central Great Barrier Reef | Geoffry Bay | M3E | 2 | 1 | 2 | 2 | 1 | 4 | 0 | 0 |
| | Nelly Bay | M4S | 2 | 1 | 2 | 3 | 2 | 4 | 5 | 5 |
| | Nelly Bay | M4E | 3 | 1 | 3 | 3 | 1 | 3 | 5 | 5 |
| | Geoffry Bay | M3S | 3 | 1 | 3 | 3 | 1 | 3 | 5 | 5 |
| | Pelurus-West | O9E | 3 | 1 | 3 | 3 | 5 | 5 | 3 | 4 |
| | Yank Jetty | O5S | 3 | 1 | 3 | 3 | 5 | 5 | 5 | 5 |
| | NE-Reef | O2E | 3 | 1 | 3 | 4 | 3 | 4 | 4 | 5 |
| | Fantome-North | O3S | 3 | 1 | 3 | 3 | 5 | 5 | 2 | 4 |
| | Florence Bay | M1S | 3 | 1 | 3 | 3 | 2 | 4 | 5 | 5 |
| | Fantomo-Northwest | O4S | 3 | 1 | 3 | 4 | 0 | 0 | 5 | 5 |
| | Pioneer Bay-North | O7S | 3 | 1 | 3 | 4 | 5 | 5 | 5 | 5 |
| | Fantome-North | O3E | 4 | 1 | 4 | 4 | 5 | 5 | 1 | 3 |
| | Florence Bay | M1E | 4 | 1 | 4 | 4 | 1 | 3 | 5 | 5 |
| | Pelurus-West | O9S | 4 | 1 | 4 | 4 | 5 | 5 | 4 | 5 |
| | Arthur Bay | M2E | 4 | 1 | 4 | 4 | 1 | 3 | 5 | 5 |
| | Pioneer Bay-North | O7E | 4 | 1 | 4 | 4 | 5 | 5 | 4 | 5 |
| | Hazard Bay | O6S | 4 | 1 | 4 | 4 | 0 | 0 | 5 | 5 |
| | Hazard Bay | O6E | 4 | 1 | 4 | 4 | 5 | 5 | 4 | 4 |
| | Yank Jetty | O5E | 4 | 1 | 4 | 4 | 5 | 5 | 4 | 4 |
| | Arthur Bay | M2S | 4 | 1 | 4 | 4 | 1 | 3 | 5 | 5 |
| | Dunk | D1S | 5 | 2 | 4 | 5 | 1 | 3 | 5 | 5 |
| | Cattle Bay | O8E | 5 | 1 | 5 | 5 | 4 | 4 | 3 | 4 |
| | Cattle Bay | O8S | 5 | 1 | 5 | 5 | 0 | 0 | 5 | 5 |
| | Iris Point | O1E | 5 | 1 | 5 | 5 | 0 | 0 | 2 | 3 |
| | NE-Reef | O2S | 5 | 1 | 5 | 5 | 0 | 0 | 3 | 3 |
| | Iris Point | O1S | 6 | 2 | 5 | 5 | 4 | 5 | 3 | 4 |
| | Fantomo-Northwest | O4E | 5 | 1 | 5 | 5 | 0 | 0 | 2 | 3 |

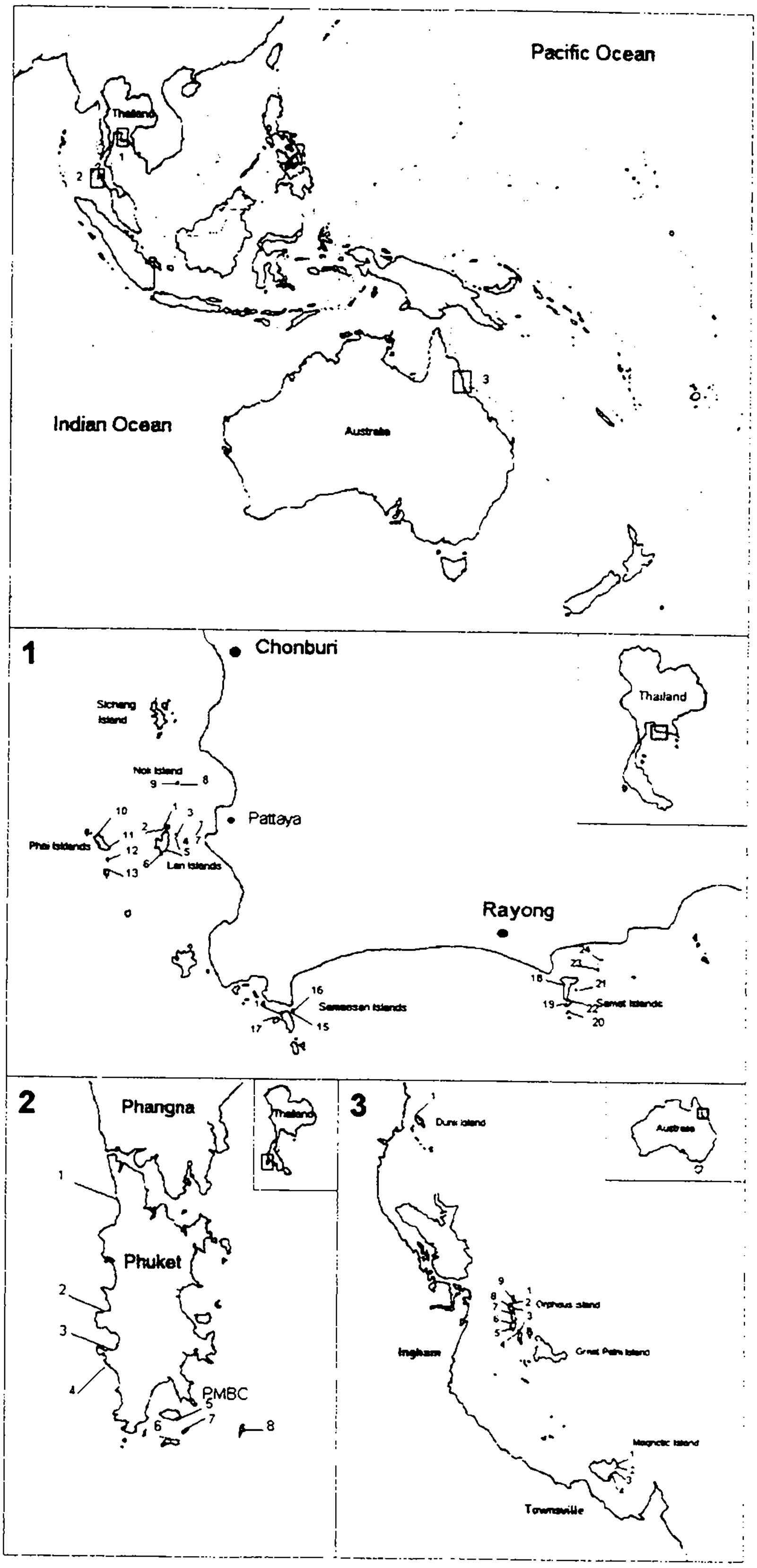


Fig. 8.1. Map showing study localities in three regions; 1) the east of the Gulf of Thailand, 2) Phuket, and 3) central Great Barrier Reef.

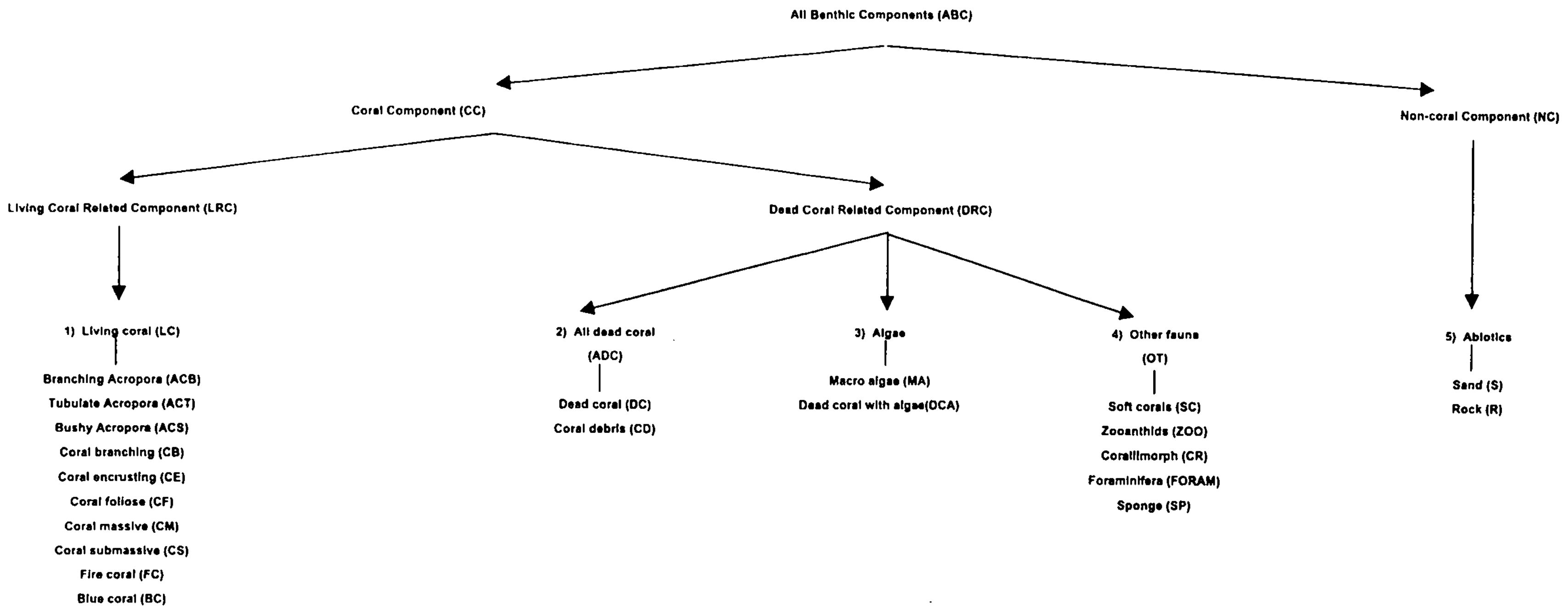


**Fig. 8.2.** Schematic diagram showing classification of the benthic lifeforms on coral reefs (21 benthic lifeforms used are modified from Dartnall & Jones, 1986).

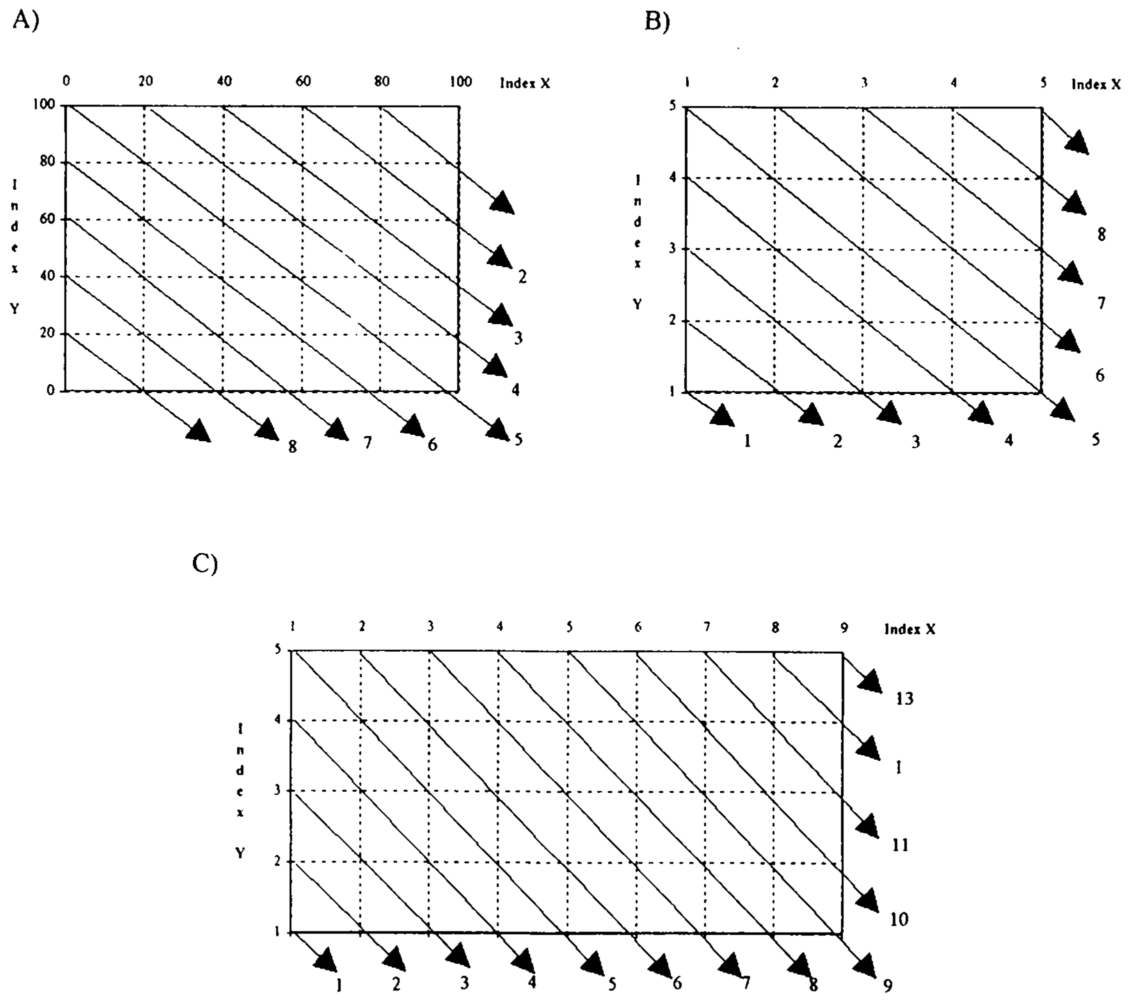
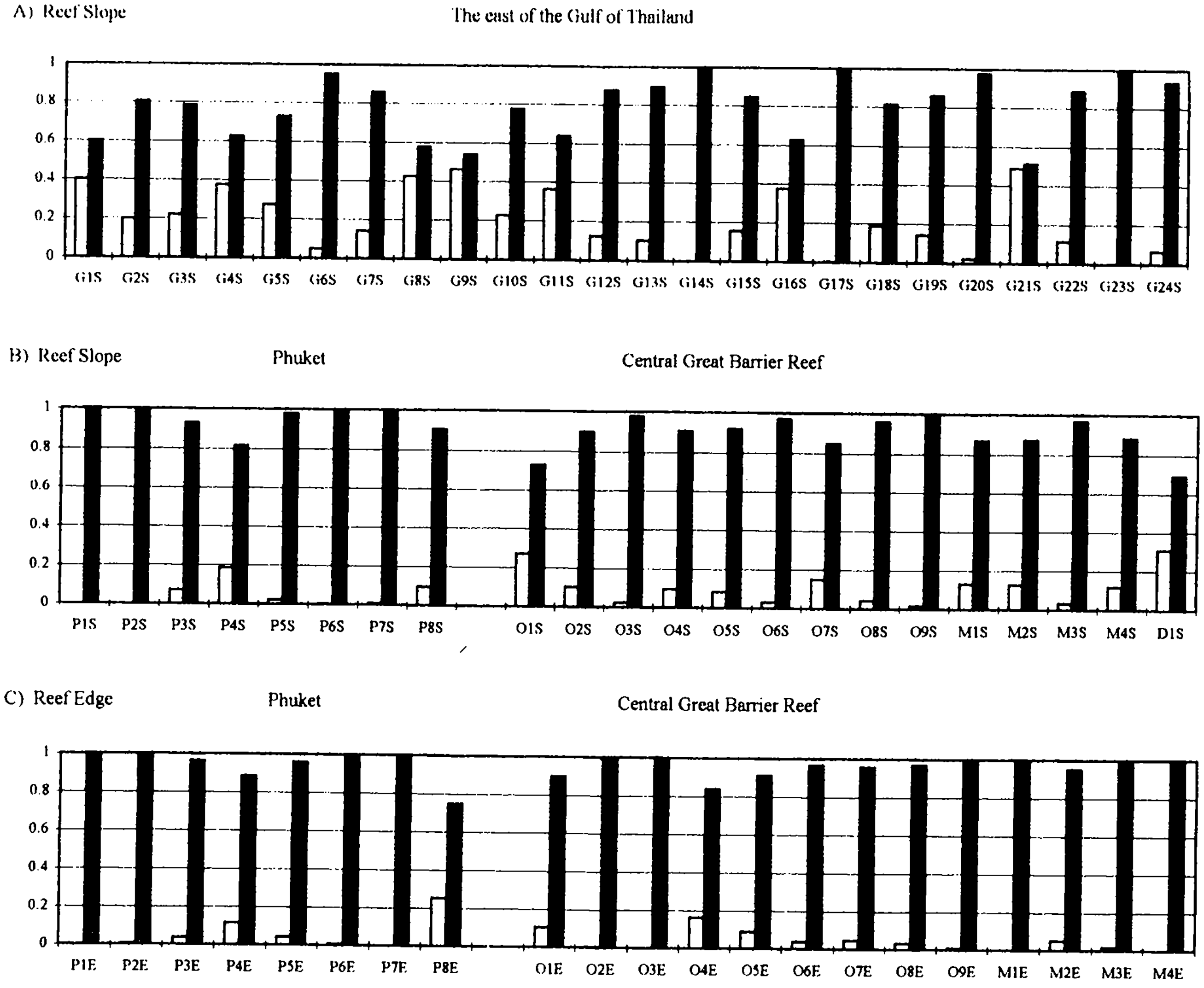


Fig. 8.3. Principle of multi-dimensional ranking method showing integrated quality ranking square of the first three dimensions. A) Primary level using percentage with balanced five classes, B) Primary level using rank quality with balanced five classes, and C) Secondary level using rank quality with non-balanced classes, Index X = 9 (result from Primary level) and Index Y = 5. Arrows are iso-ranking lines and numbers beside each arrow indicate new ranking quality. Sites within each grid of A) that an arrow passes through are classified as the same rank while in B) and C), site will locates on the intersect.



A) Reef Slope The east of the Gulf of Thailand

B) Reef Slope Phuket Central Great Barrier Reef

C) Reef Edge Phuket Central Great Barrier Reef

Fig. 8.4 Bar charts show the results of non-coral related component area cover (blank column) and development index (fill column). A) the east of the Gulf of Thailand, B) reef slope of Phuket and central Great Barrier Reef, and C) reef edge of Phuket and central Great Barrier Reef. The details of the site code name are as follow; the first letter indicate localities (G-the east of the Gulf of Thailand, P-Phuket, O-Orpheus Islands, M-Magnetic Island, D-Dunk Island), second number indicate locality number and third letter indicate habitats (S-reef slope and E-reef edge). Full site names are presented in Table 8.4.

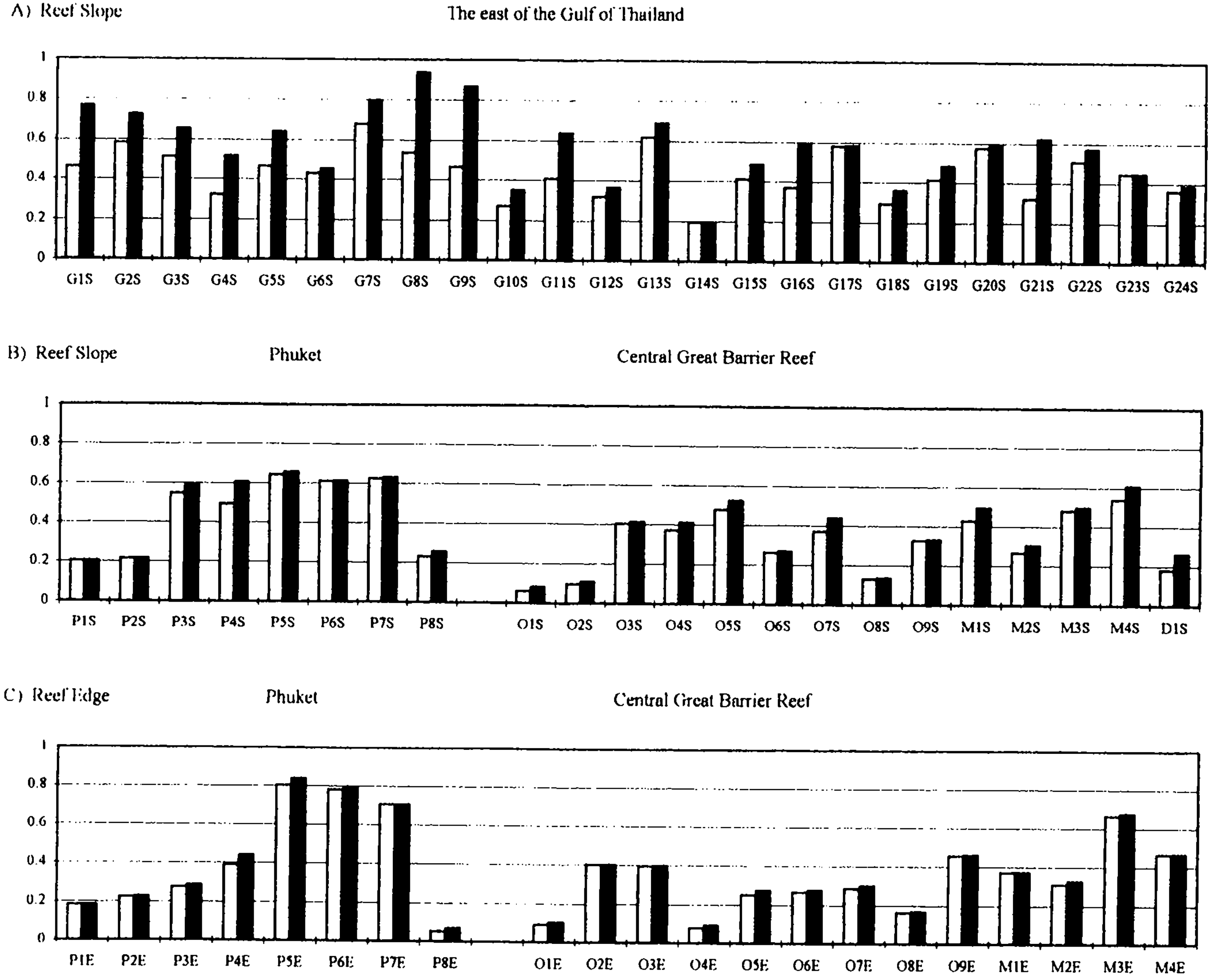
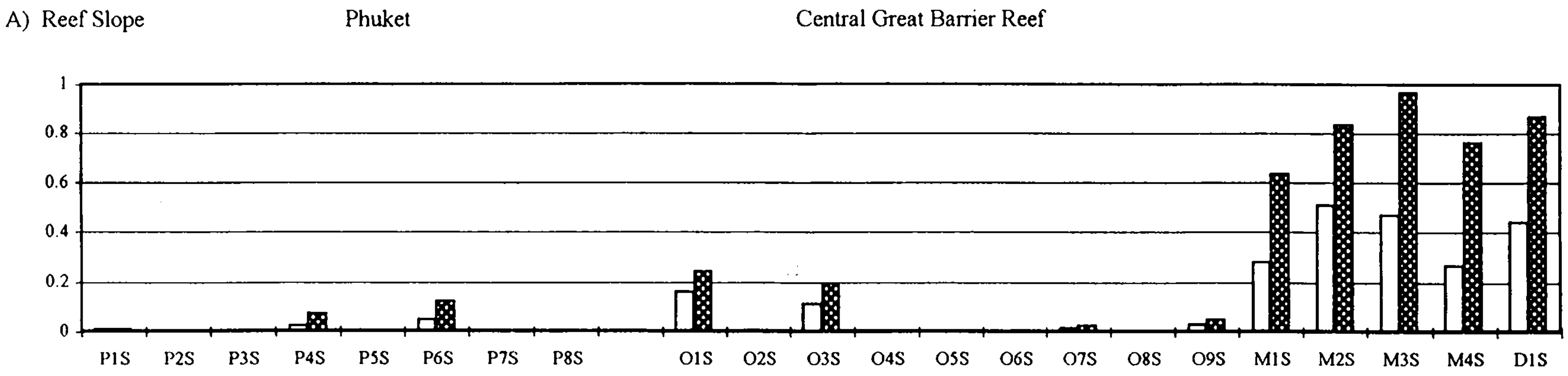


Fig. 8.5. Bar charts show the results of living coral area cover (blank column) and condition index (fill column): A) the east of the Gulf of Thailand, B) reef slope of Phuket and central Great Barrier Reef, and C) reef edge of Phuket and central Great Barrier Reef. The details of site code name are as in Fig. 8.4 and Table 8.4.



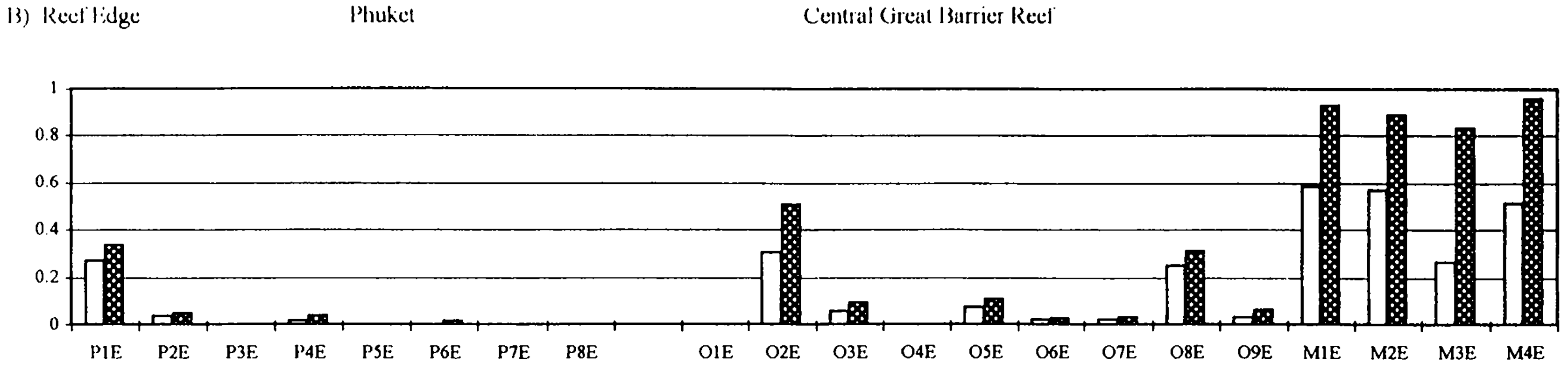


Fig. 8.6. Bar charts show the results of algae area cover (blank column) and algal index (fill column). A) reef slope of Phuket and central Great Barrier Reef, and B) reef edge of Phuket and central Great Barrier Reef. The details of the site code name are are as in Fig. 8.4 and Table 8.4.

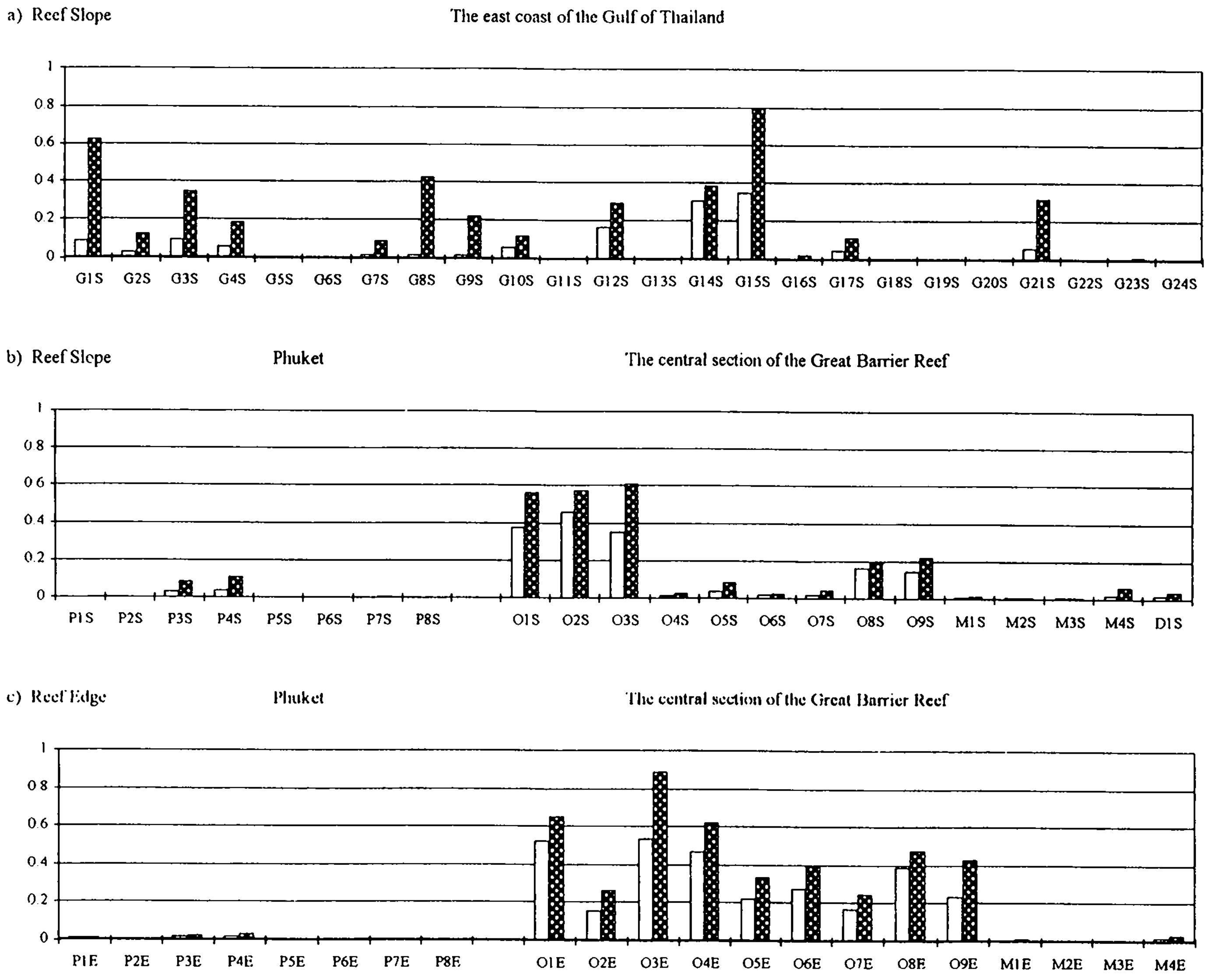
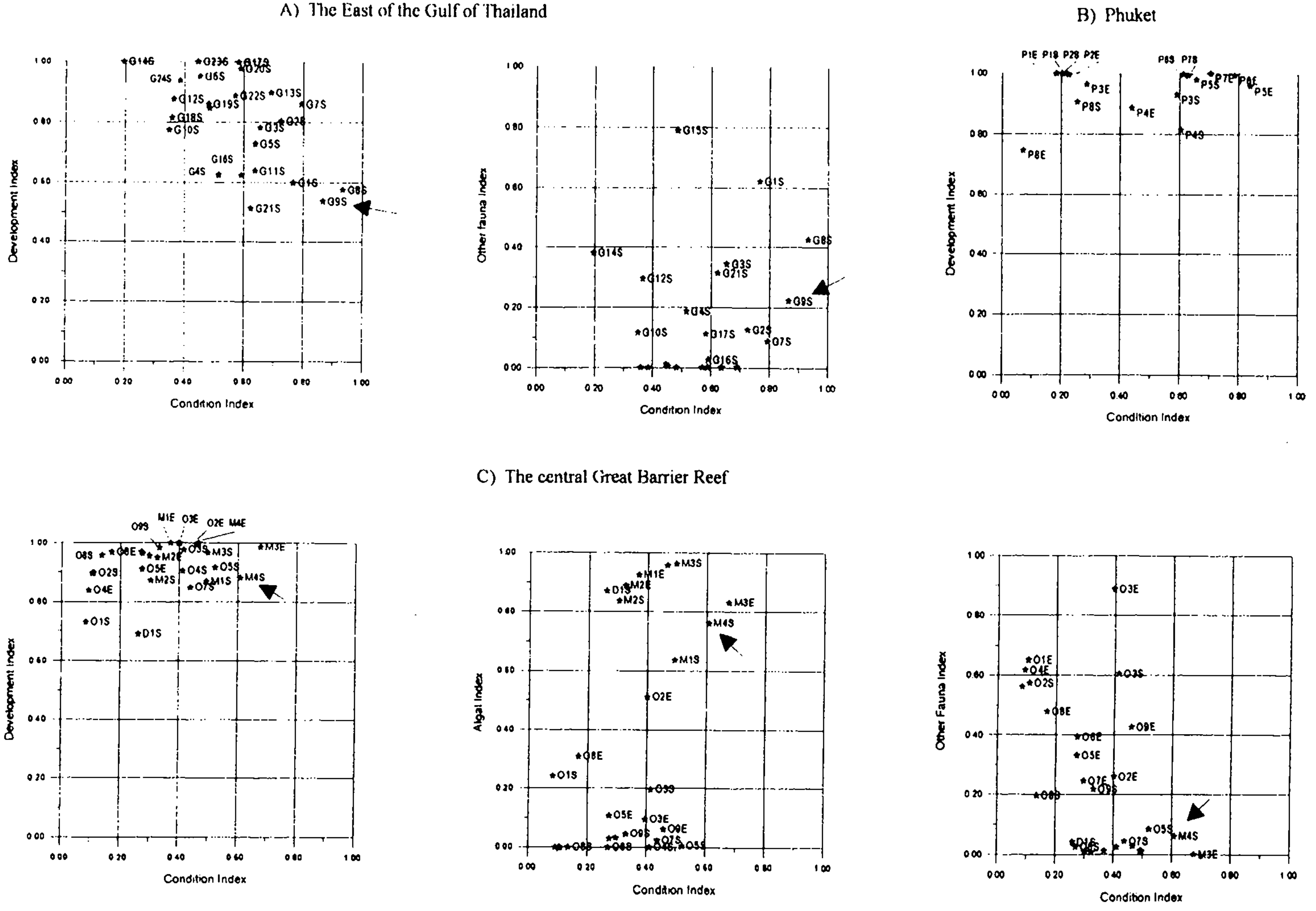


Fig. 8.7. Bar charts show the results of other fauna area cover (blank column) and other fauna occupying index (fill column). A) the east of the Gulf of Thailand, B) reef slope of Phuket and Central Great Barrier Reef, and C) reef edge of Phuket and central Great Barrier Reef. The details of the site code name are as in Fig. 8.4 and Table 8.4.



A) The East of the Gulf of Thailand

B) Phuket

C) The central Great Barrier Reef

Fig. 8.8. Scatter diagrams of the study sites plot between Condition Index and other three indices: Development, Algal, and Other Fauna Indices. A) The east of the Gulf of Thailand, B) Phuket, and C) Central Great Barrier Reef. The diagrams of each region are not all shown because the significant of each index in each region. Sites label are not all shown because overcrowded problem. Arrows indicate sites which which are described in the text.

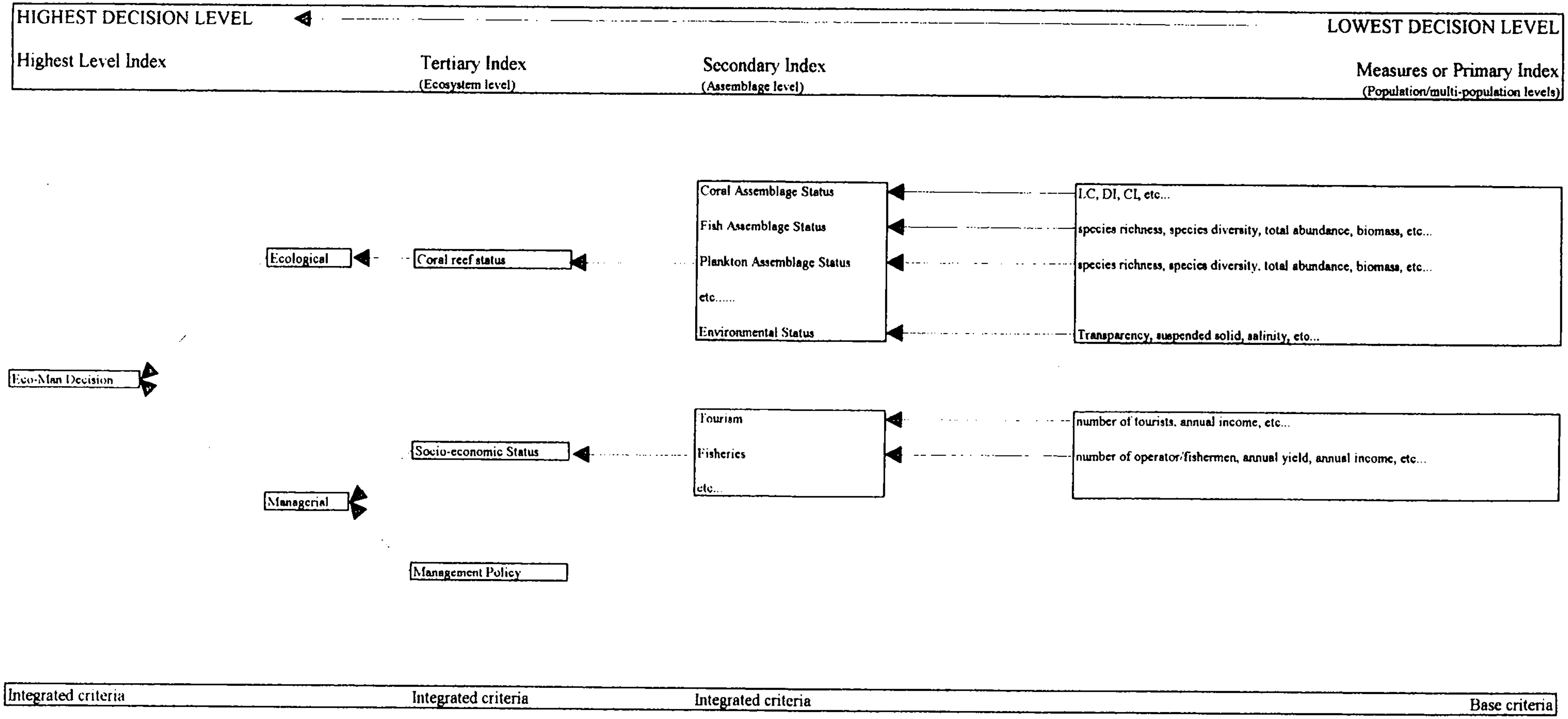


Fig. 8.9. Diagram showing multiple decision levels model for coral reef resources management based on different levels of information.

# CHAPTER 9

# GENERAL DISCUSSION

This thesis has confirmed that habitat structure plays an important role in influencing the structure of fish assemblages on coral reefs in two geographic regions. While the specific relationships varied, depending on the region, the locality or reef zone studied, the family or species of fish in question, and the habitat characteristics that were measured, a number of predictable patterns emerged. Although relationships between fish populations (or assemblages) and habitat structure clearly exist, quantitative relationships may or may not be detected, because of a number of factors, including the sampling design, the parameters measured and analytical techniques employed (see review in Sale, 1991a). In the first part of this discussion, I review the problem of designing sampling programs to detect and measure these relationships. In the second part, I focus on the general patterns to emerge from quantitative sampling, by measuring community structure using different parameters. In the third section, I discuss the links between fish community and habitat characteristics that have been demonstrated by experiment, contrasting the effects of decreasing and increasing the availability of different habitats. The fourth section examines regional processes that may have influenced the local patterns observed, and in part 5, I make recommendations for future work, which must focus on the demographic mechanisms underlying fish-habitat relationships and the role of resource-limitation. Lastly, I discuss the use of habitat characteristics and decision-making criteria for assessing the status of coral reefs.

## 9.1 SAMPLING DESIGN

To date, little attention has been given to developing sampling strategies to detect and measure fish-habitat relationships. In this study, the effect of different sampling schemes on the quantitative relationship between fish assemblages and habitat structure were examined (Chapter 3). The main result was that the sampling scheme which provides the most precise estimates of both fish assemblages and habitat structure, will provide the most consistent and precise quantitative relationship between the two. An optimised sampling scheme that combines fish and habitat data is recommended for all future studies addressing this problem, so that conclusions about the importance of habitat characteristics can be made from reliable information. Optimization of a sampling scheme for censusing multiple species or whole assemblages could also be conducted by applying the concept of cost efficiency (e.g. Pringle, 1984; Andrew & Mapstone, 1987). The results of this study indicate that quantitative relationships between fish and habitat components are substantially influenced by the sampling technique, transect dimensions and replication.

In addressing fish-habitat relationships it is critical that data for both fish assemblages and habitat structure should be collected at the same place, same time and same sampling units. This is important as most reef fishes usually inhabit a particular reef habitats (Chapter 4 and 5 and also Russ, 1984a, 1984b; Meekan *et al.*, 1995; Green, 1996) and both fish and microhabitats are patchily distributed within broad habitat zones. Both fish assemblages and habitat structure can vary over a number of temporal scales (Williams, 1983b). Some previous studies have overlooked this problem (e.g. Roberts & Ormond, 1987).

Line transect techniques provide an appropriate method to simultaneously record fish abundance and habitat information, and allow fine-scale spatial and temporal patterns of association to be described. In this study, the combined application of the instantaneous visual census technique for fish assemblages (Fowler, 1987) and the line intercept transect technique for habitat structure (Loya, 1978; English *et al.*, 1994), provides a cost efficient method of obtaining both types of data. Therefore, this sampling scheme has the potential to become a standard for reef monitoring programs.

## 9.2 FISH-HABITAT RELATIONSHIPS

Most of previous studies investigated fish-habitat relationships by considering linear relationships or trends based on univariate descriptions of fish communities (e.g. species diversity) and habitat structure (e.g. % coral cover or rugosity) (e.g. Luckhurst & Luckhurst, 1978; Bell & Galzin, 1984; Roberts & Ormond, 1987; McClanahan, 1994; Green, 1996). This approach places limits on the types of pattern that will be detected. The use of summed variables and indices can result in a loss of information, making it less likely that effects will be detected. The assumption that relationships, if they exist, will be linear, can also be misleading. That is, an existing relationship may be interpreted as no relationship at all. There is increasing information (including this study) to suggest that relationships between two or more components on the reef are likely to be complex non-linear relationships (e.g. McClanahan, 1994; Meesters *et al.*, 1996).

In this study, investigations were carried out using both univariate and multivariate approaches (Chapter 4 and 5). The conventional univariate approach was applied, but with the intention of detecting and describing both linear and polynomial relationships. Another approach was carried out with the aid of multivariate statistical analyses, to explore more detailed relationship between fish assemblages and habitat structure. These techniques are considered mores sensitive indicators of patterns of change in the composition (community structure) of each component, and can be used to identify concordant patterns.

The detected associations between fish community parameters and major habitat structure included both linear and parabolic relationships. The type of relationship varied, depending on the fish family examined, the habitat attributes measured, the parameters considered, and the place and time of the study. In both regions studied (Thailand and Australia), labrids showed linear relationships between community measures such as species richness and habitat structure. The same result can be seen for species diversity which exhibited a negative linear relationships with living hard coral cover. Not all the results for parameters (pairs) were statistically significant, but they usually exhibited the same trend. Species diversity, for example, tended to have a positive relationship with dead coral and soft coral. However, the relationship between labrids and habitat structure might not be the case for other regions e.g. Lizard Island, northern GBR (Green, 1996).

For pomacentrids and chaetodontids, which were only studied at Phuket, there were weaker relationships between univariate community measures and major attributes of the habitat. While fish species diversity was found to exhibit linear relationships with major benthic area cover, species richness and evenness index appeared to exhibit reciprocal parabolic relationships.

The quantitative relationships between fish and habitat assemblages varied over time, suggesting static sampling designs may provide an incomplete picture of the strength of fish-habitat linkages. However, generally where temporal changes occurred they did not alter or obscure the spatial pattern in the data. Locations exhibiting a large change in habitat structure also experienced a large change in the characteristics of the fish assemblage (e.g. 3 locations at Phuket). Where habitat structure remained fairly static, fish assemblages appeared to persist (e.g. Orpheus Is). This may not be true for other locations or habitats (see Sale *et al.*, 1994).

The importance of habitat structure for associated fish depends upon the resource(s) exploited from the habitat, be it (they) living space, shelter and/or food (Sano *et al.*, 1987; Hixon & Beets, 1993). The nature of relationship is, therefore, primarily a predictive role for habitat structure and a responsive role for fish (unless fish actively modify habitat structure). However, from the point of view of fish as indicators of change in habitat structure, the reverse applies. That is, the ability to predict habitat changes from information on the fish fauna is important. This study shows that, in statistical terms, there can be an asymmetry in our ability to predict fish from habitat and vice versa. The nature of this asymmetry varies among groups of fish. For example, labrids have a predictive role, chaetodontids have a responsive role while pomacentrids have both. It is likely that these patterns are dynamic and will change over different spatial and temporal scales. The processes or factors underlying these contrasting patterns are fully not understood, but the relative commonness of both components (fish and habitat variables) and the degree of specialization by fish will no doubt be important.

For example, if *Labicthys unilineatus* presented, it was possible to predicted that branching *Acropora* is also dominate in that area. In contrast, if branching *Acropora* dominated on the reef, ability to predicted that *Labicthys unilineatus* is also dominated or presented in that reef is lower than in the first case.

## 9.3 EXPERIMENTAL MANIPULATIONS

Experimental manipulations provided an opportunity to test the influence of the availability of different types of habitat on fish assemblages, by both reducing and increasing the availability of resources. The results gained from experimentally increasing resources in depleted areas also enabled the potential effectiveness of habitat restoration to be tested (Jordan, 1995).

The results from the observational chapters suggested that relationships between fish and habitat varied among fish families and species. Overall, all fish assemblages contained species associated with different substrata, with perhaps the wrasses exhibiting the greatest range and strongest habitat associations. For example, there are labrid species associated with live coral, dead coral, macroalgae and soft coral. In this study, the influence of particular types of major habitat structure were tested in terms of habitat degradation and rehabilitation. The scale of the experiment, however, was tested at the lower half (0-50%) of area cover of target habitat structures. At this scale, any existing quantitative relationship appears to be linear even if it is quadratic over the full range (0-100%) of habitat cover (see Chapter 4 and 5).

In general, fish responded to habitat (resource) alteration in predictable ways, for both degradation and rehabilitation experiments. For example, hard coral associated species increased when habitat was enhanced and declined when habitat was disturbed. However, there are many interactions which result from removing one structure (e.g. coral), which increases another (e.g. dead coral). For habitat degradation, when living coral was removed, a negative effect was detected on fish that associated with living corals such as pomacentrids. However, removing living coral provided more "bare space" which was utilized labrids. A positive effect on some labrids were therefore detected in this experiment. Similar results were also detected for the removal of soft corals. Removal of soft coral had a positive effect in terms of (increase) habitat complexity. Some fish, therefore, recolonised the new available space (both from migration and recruitment). Patterns observed at the community level appeared to reflect the balance of individual responses among the constituent species.

These experiments were carried out on fringing reefs and the experimental units were small, and not isolated from neighbouring habitats. Migration of fish from surrounding areas, therefore, may have been an important consideration in the interpretation of results. Spatial differences of reef condition, at both local and regional scales, also had an influence on the results. The relatively non-disturbed reefs (Orpheus Island) had a greater availability of potentially "source" fish assemblages, while the opposite was true for the highly disturbed reefs of Maiton Island. The magnitude of change is also important and relevant to the strength of relationship between fish and habitat. The strength of relationships can be considered from the slope of the models relating fish and habitat. A quick response from fish to experimental changes could be expected from a sharp slope.

The application of these results for management in terms of improving reef condition to promote fish assemblages, should be undertaken with caution. The type and magnitude (e.g. % cover) of habitat structure introduced should be considered in relation to the types of fish being targeted. In increasing a particular structure, the optimal magnitude of the increase will need to be considered. The removal of some habitat structures can also give a positive results for fish e.g. soft coral. Reef rehabilitation does not therefore have to consist of the introduction of some habitat types, but can also involve the removal of some structures. The introduction of artificial habitat may be useful in some cases.

The concept of reef rehabilitation or enhancement should be considered carefully. This concept may be different for artificial reefs, where the main objective is to promote fish production. Existing plans for using artificial reefs generally consider utilizing areas adjacent to natural habitat (e.g. Ambrose & Swarbrick, 1989). Rapid colonization by fish moving off natural habitat does not increase fish production or abundance in the short term (Bohnsack, 1989). This might create further overfishing problems as adult populations in natural habitat may already be caught and the few remaining fish will move to the artificial reef where they can be easily caught. Application should therefore focus on promoting production rather than attraction alone (Grove & Wilson, 1994).

For coral reef rehabilitation, objectives would primarily be to improve benthic assemblages (habitat structure) and then associated assemblages i.e. fish (Clark & Edwards, 1994). If degraded reefs have the potential to recover by themselves it might not be necessary to manipulate habitat structure and only minor improvements may be enough. Fast recovery of benthic and fish assemblages on coral reefs has been reported elsewhere, especially when no further disturbance occurs (Pearson, 1981; Colgan, 1987; Done *et al.*, 1988). Habitat improvement may therefore only be necessary for reef that have a low ability for self recovery. These reefs may be significant in terms of ecological (e.g. connection of reef organisms to a larger reef system) or socio-economic (e.g. fishing area for locals, tourist attraction area). This is important because any activities involving habitat improvement are very expensive, and doing the right thing at the right place means cost effective resource management.

## 9.4 REGIONAL COMPARISONS

The results of observational studies indicate that both regions (Thailand and Australia) share the common feature in that reefs are dominated by pomacentrids and labrids, although there are considerable differences in the regional species pools. Other families such as herbivorous-grazers (e.g. scarids, acanthurids, siganids) and coral feeders (e.g. chaetodontids) play a minor role in fish assemblages at both Maiton and Orpheus Islands. Both these areas can be considered as near-shore reefs where a lower representation of the latter groups have been reported (Williams, 1982; Williams & Hatcher, 1983, Russ, 1984a, 1984b).

There were some differences in reef condition between regions in terms of both fish assemblages and habitat structure. At Maiton Island, labrids and pomacentrids were found in equal proportions within the fish assemblages as a whole, while habitat structure was heavily degraded. In contrast, there was no sign of dramatic physical damage on reefs at Orpheus Island, while fish assemblages were dominated by pomacentrids. This difference might be explained by the intermediate disturbance hypothesis (Connell, 1978). This hypothesis predicts that assemblages in highly disturbed environments will be dominated by a few resistant species with good colonizing abilities, which are gradually replaced by competitively superior, but more sedentary species as stability increases (Connell, 1978; Death, 1995). At intermediate levels of disturbance, a greater diversity of species can persist, as inferior competitors are not excluded. The highly disturbed reefs at Maiton Island may be more susceptible to the invasion of labrid fishes, which have good colonizing ability, and because they are associated with dead coral surfaces, are promoted by disturbance.

The experiment on both region were carried at different time and they may also at different succession stage. Because of the experiments were carried out within a limited period of time which might not cover all of the succession process. Long term study on the influence of habitat structure changes on reef fish assemblage is necessary.

## 9.5 FUTURE DIRECTIONS

While a strong linkage between reef fishes and habitat structure has been described, the demographic processes responsible for these patterns remain to be determined. Patterns may be established at the time of settlement (Doherty *et al.*, 1996) or throughout the post settlement, reef-associated phase of the life cycle (see review in Jones, 1991). While the relationships between fish assemblages and habitat structure could be explained by a conventional resource-limitation model, this need to be further tested by larger-scale experiments. Results suggest that species closely associated with a particular resources may respond differently, depending on whether or not the resource is increasing or decreasing (e.g. Wellington & Victor, 1985). Habitat-limited fishes are likely to exhibit a decline in abundance when the resource declines. However, fish may not necessarily respond to an increase in the availability of preferred habitat, particularly when other species are exploiting the habitat or habitat becomes gradually less limiting above a threshold level. Future work must focus on the relationships between individual species, their competitors and the availability of resources.

## 9.6 HABITAT STRUCTURE AND MANAGEMENT

Ecologists have a responsibility to supply decision making tools especially from biological/ecological understanding. In the last part of this study I proposed a decision making framework or assessing the status of reefs on the basis of benthic assemblages (Chapter 9). This kind of integrated approach is likely to play an important role in present and future of resource management plans, especially through Geographical Information Systems (GIS). The relationships between variables may also play a vital role in this process, as the relationships between variables are not always simple linear relationships, but tend toward complex non-linear relationships (Meesters *et al.*, 1996). The logic behind the decision model may be used to overcome this problem. In general, the decision has been based on conventional Boolean's logical theory which has a clear cut threshold for acceptance or rejection. This theory is not suitable for diverse relationships between variables. More recently, the Fuzzy logic theory (Zadeh, 1965, 1990) and multi-criteria approach have become popular as management decision making tools especially in GIS (Banai, 1993). This method has only very recently been applied to coral reef ecosystems (Meesters *et al.*, 1996). The procedure does not have strict decision thresholds and therefore has the flexibility to cope with uncertain relationships of multiple variables. However, the development of such a decision making tools, based on the different logical theory, was beyond the scope of this study.

## 9.7 CONCLUSIONS

In conclusion, the study has been one of the first to attempt a biogeographic synthesis of the relationship between coral reef fish and the complex habitat they occupy. Both observational and experimental methods combined to detect and describe a number of general patterns that are important in both ecological and management terms. A sound knowledge on their relationships can lead to an increased understanding of the factors limiting populations and structuring fish assemblages. This knowledge can also be utilized in attempts to achieve sustainable use of coral reef resources, by providing better choices for ecological indicators, information on responses to anthropogenic disturbance, information on the effectiveness of habitat restoration techniques and decision-making tools that can be used in status assessment and the selection of sites for marine reserves. The thesis lays the foundation for future studies which must focus on the mechanisms that constrain fish communities within bounds determined by their underlying habitat.

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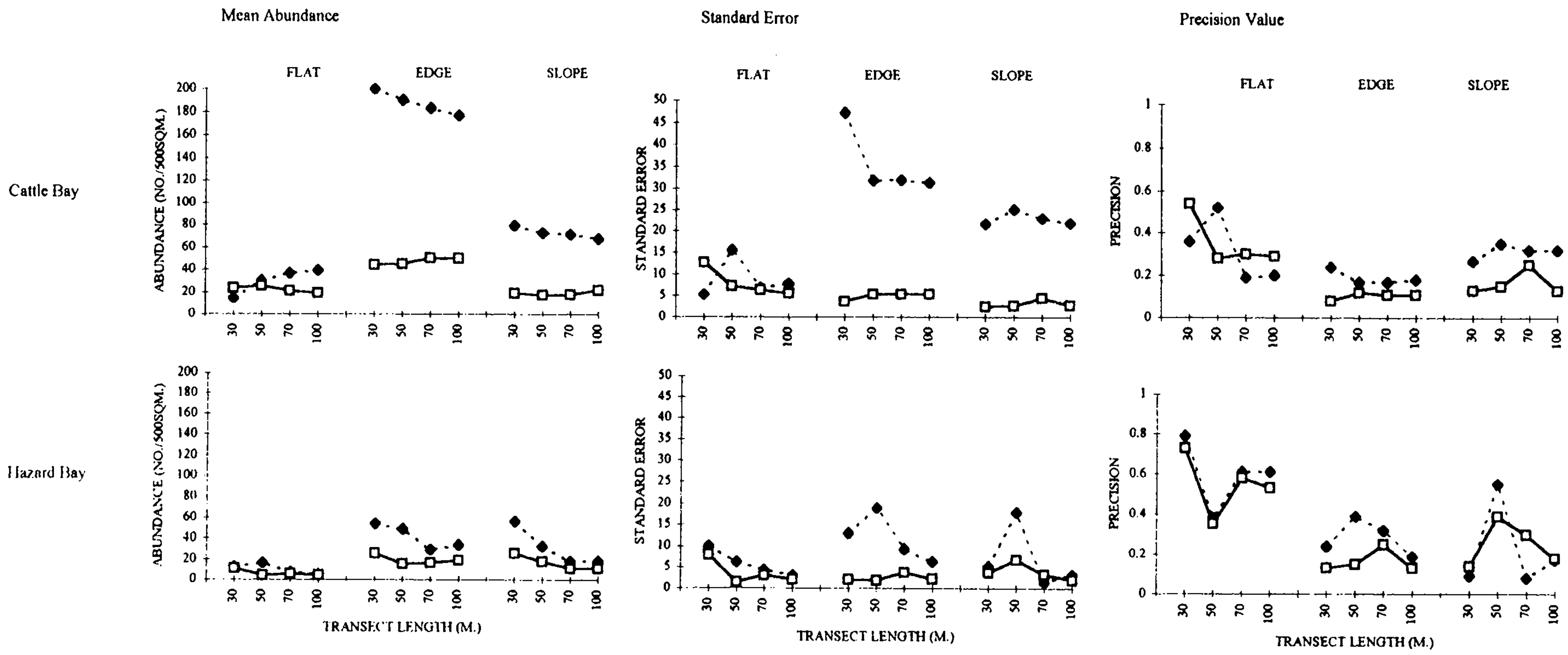
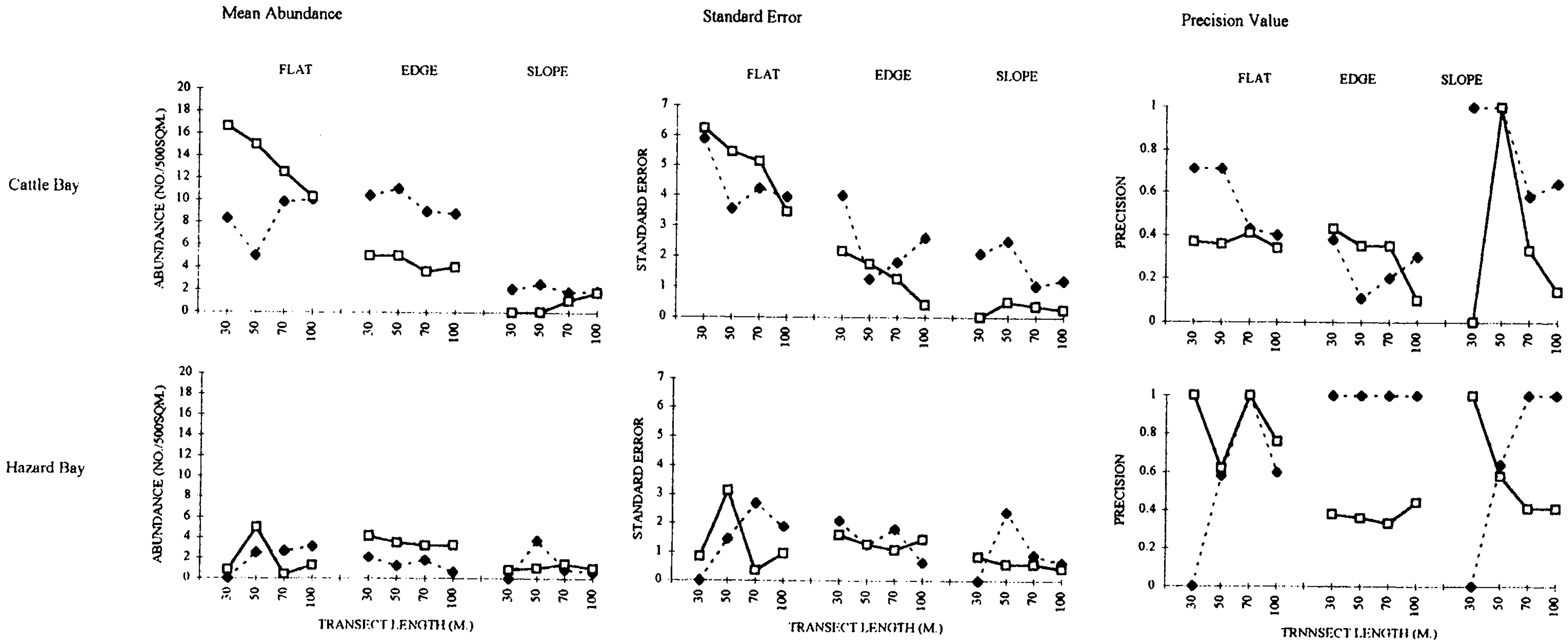
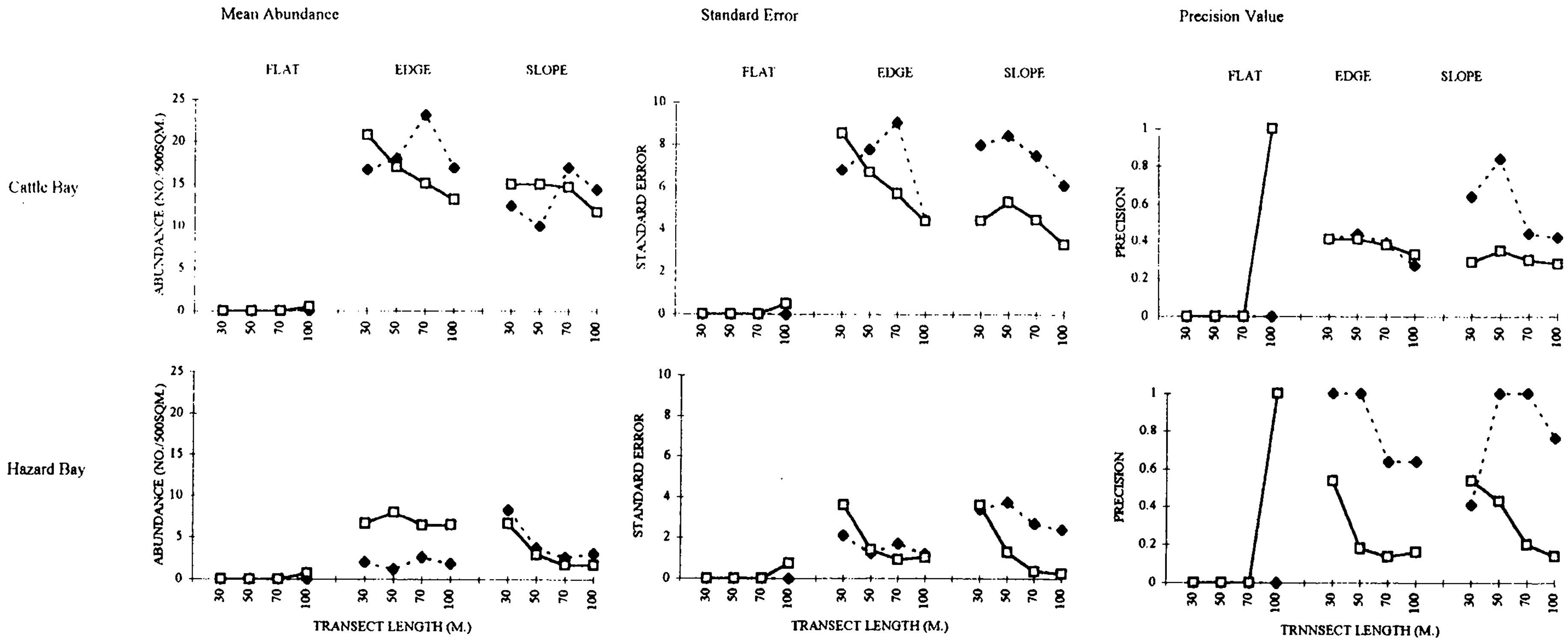
# APPENDICES



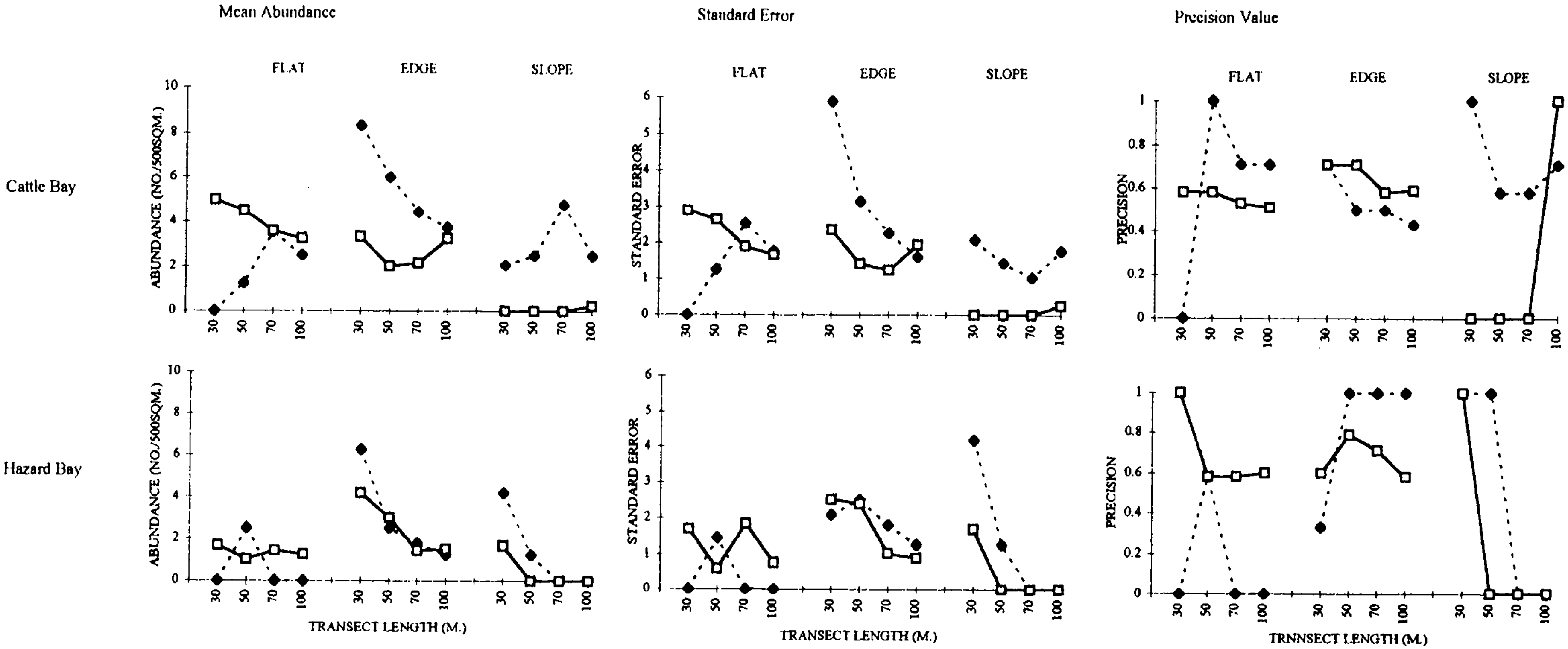
Appendix 3.1. Results of the transect dimension study estimating *Halichoeres melanurus* at Cattle Bay and Hazard Bay. Graphs show Mean Abundance, Standard Error and Precision Value: ◆----------◆ 2m wide transect, □– ——□ 5m wide transect.



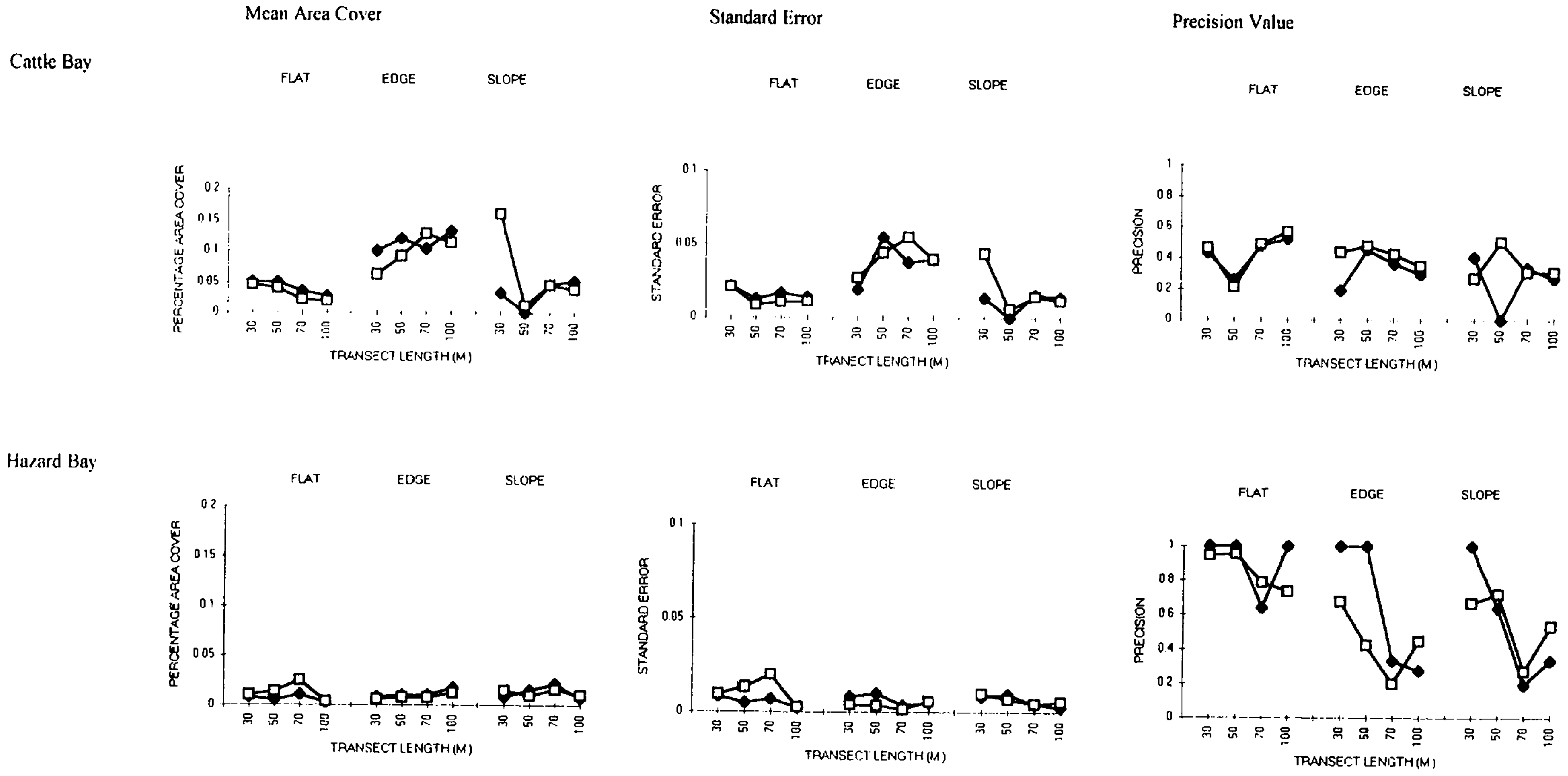
Appendix 3.2. Results of the transect dimension study estimating *Halichoeres chloropterus* at Cattle Bay and Hazard Bay. Graphs show Mean Abundance, Standard Error and Precision Value; ◆---------◆ 2m wide transect, ☐ ☐ 5m wide transect.



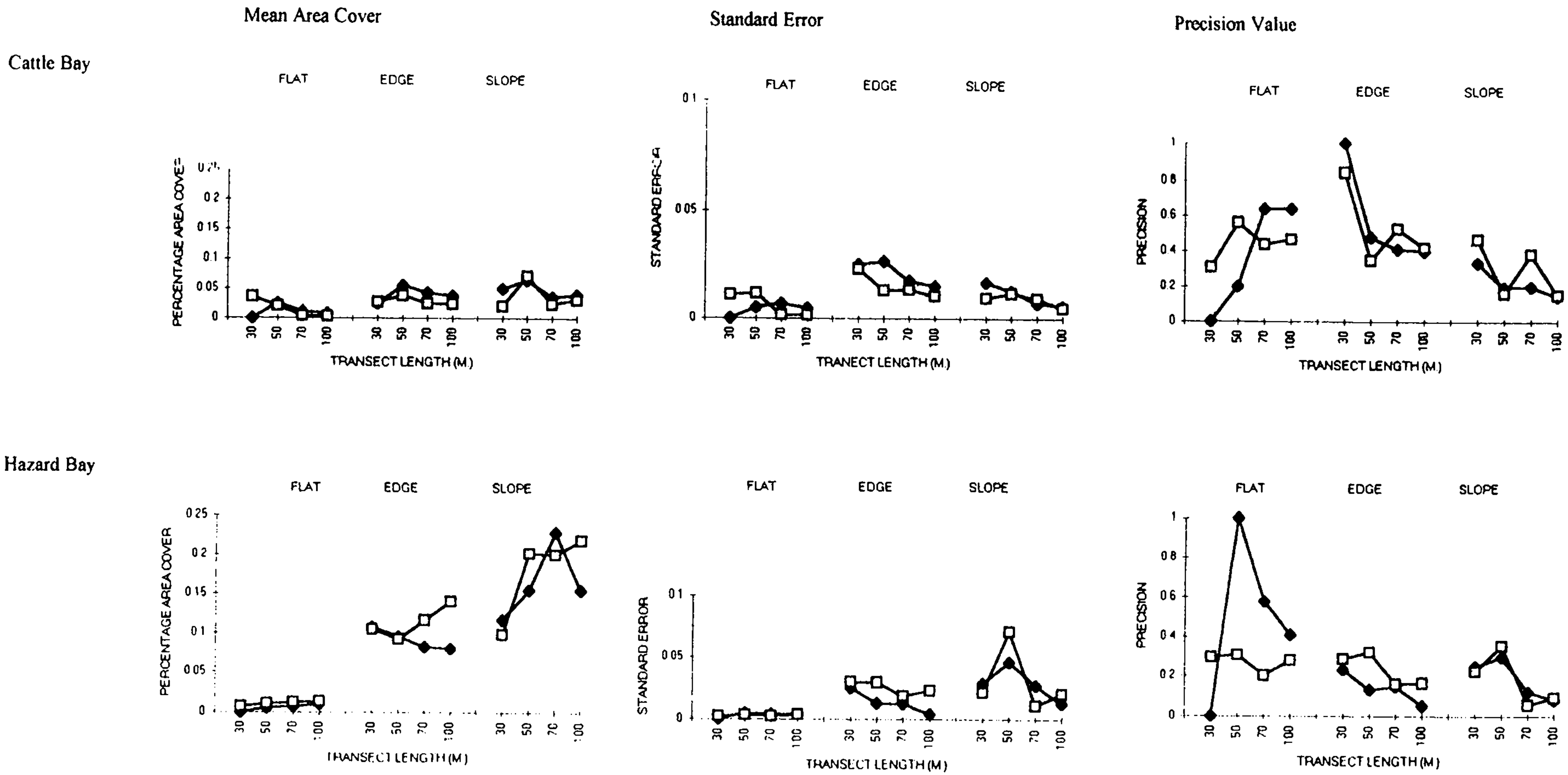
Appendix 3.3. Results of the transect dimension study estimating *Thalassoma lunare* at Cattle Bay and Hazard Bay. Graphs show Mean Abundance, Standard Error and Precision Value: ◆-----------◆ 2m wide transect, ☐ ― ☐ 5m wide transect.



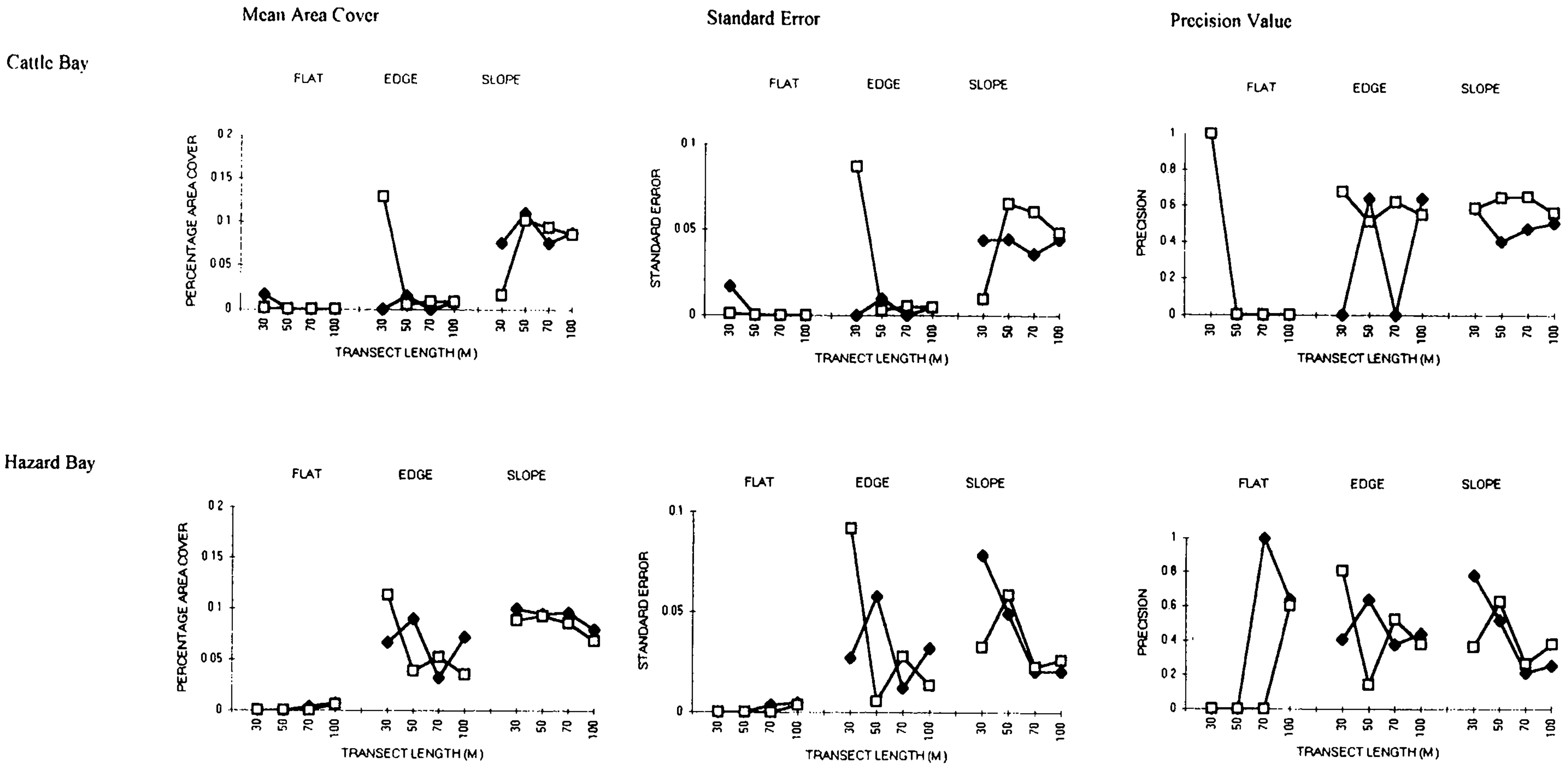
Appendix 3.4. Results of the transect dimension study estimating *Stethojulis strigiventer* at Cattle Bay and Hazard Bay. Graphs show Mean Abundance, Standard Error and Precision Value: ◆---------◆ 2m wide transect, □— —□ 5m wide transect.



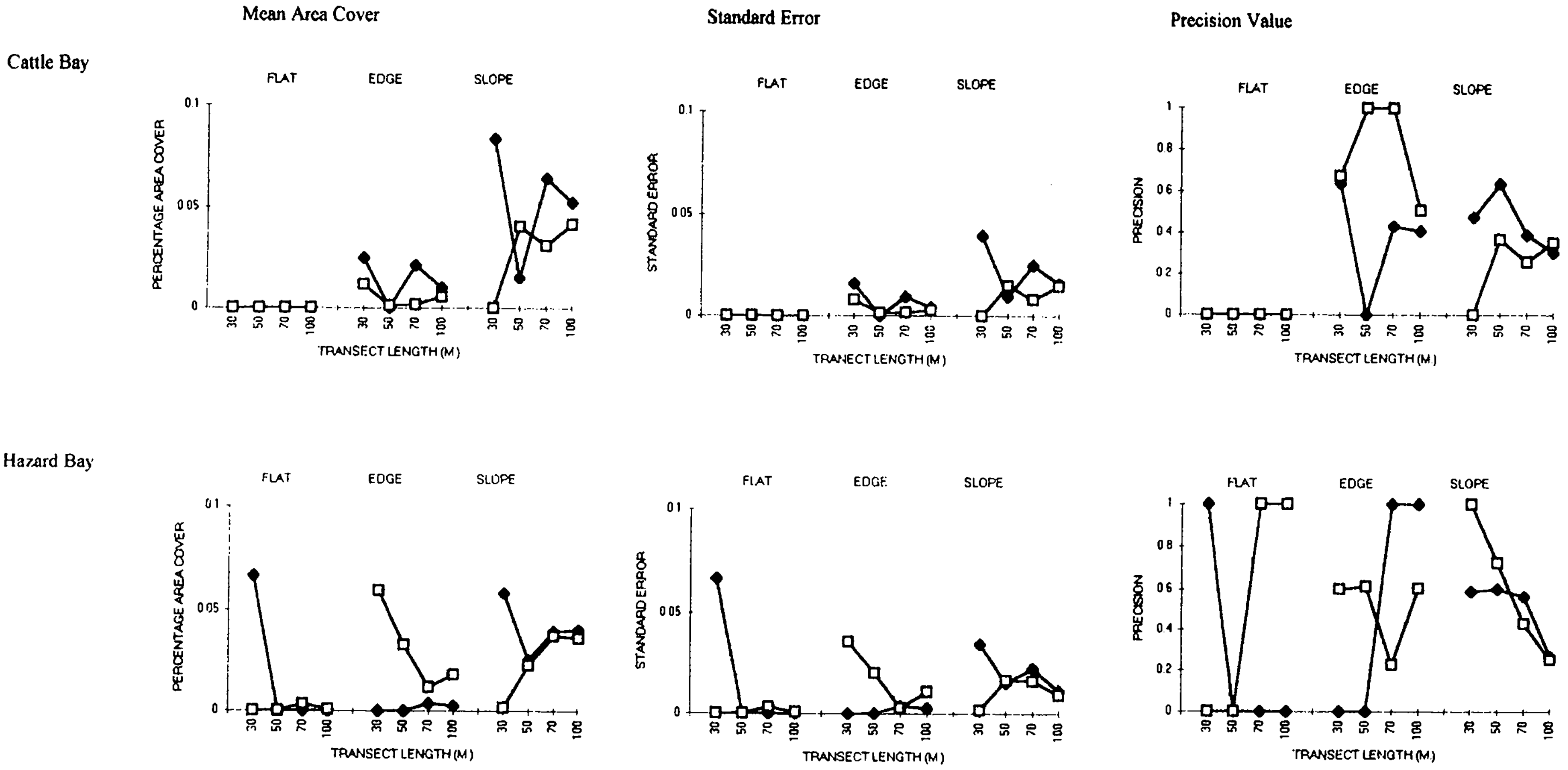
Appendix 3.5. Results of the sampling technique study estimating *Acropora* Coral (AC) at Cattle Bay and Hazard Bay. Graphs show Mean Area Cover, Standard Error and Precision Value; ◆------◆ Fixed Density Point Transect □------□ Line Intercept Transect.



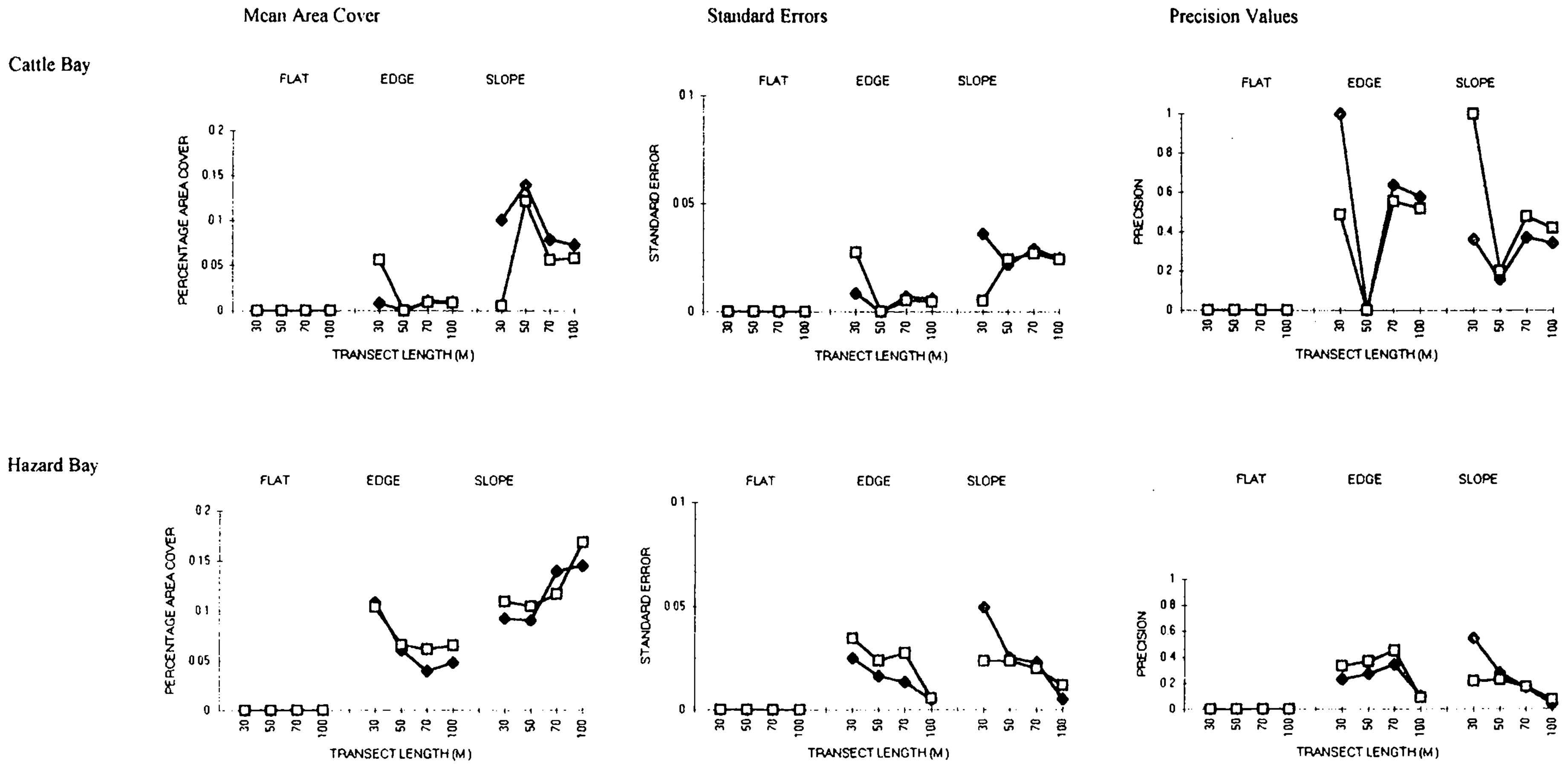
Appendix 3.6. Results of the sampling technique study estimating Massive Coral (CM) at Cattle Bay and Hazard Bay. Graphs show Mean Area Cover, Standard Error and Precision Value; ◆-------◆ Fixed Density Point Transect ☐------☐ Line Intercept Transect.



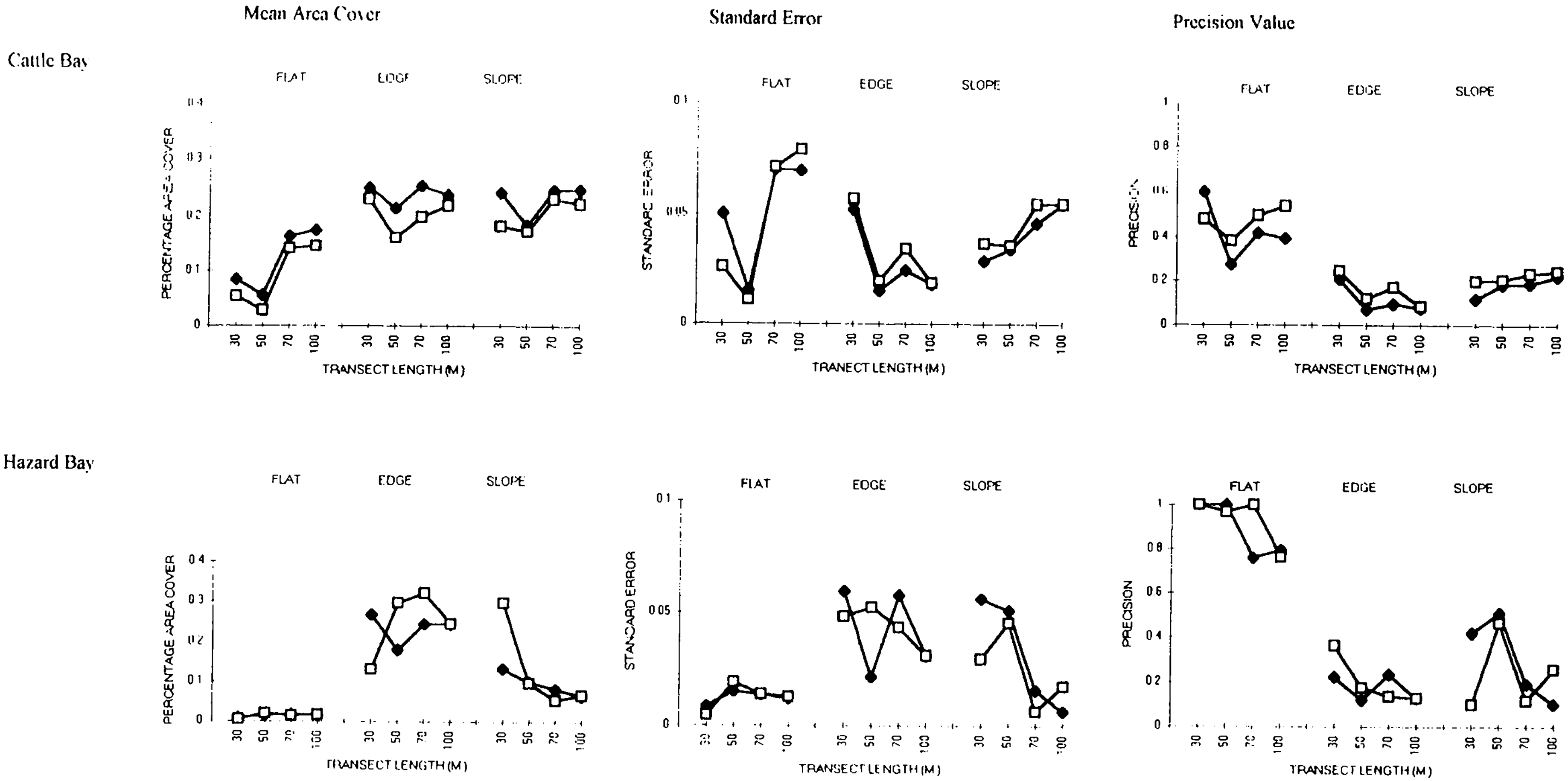
Appendix 3.7. Results of the sampling technique study estimating Branching Coral (CB) at Cattle Bay and Hazard Bay. Graphs show Mean Area Cover, Standard Error and Precision Value; ◆-----◆ Fixed Density Point Transect, ☐------☐ Line Intercept Transect.



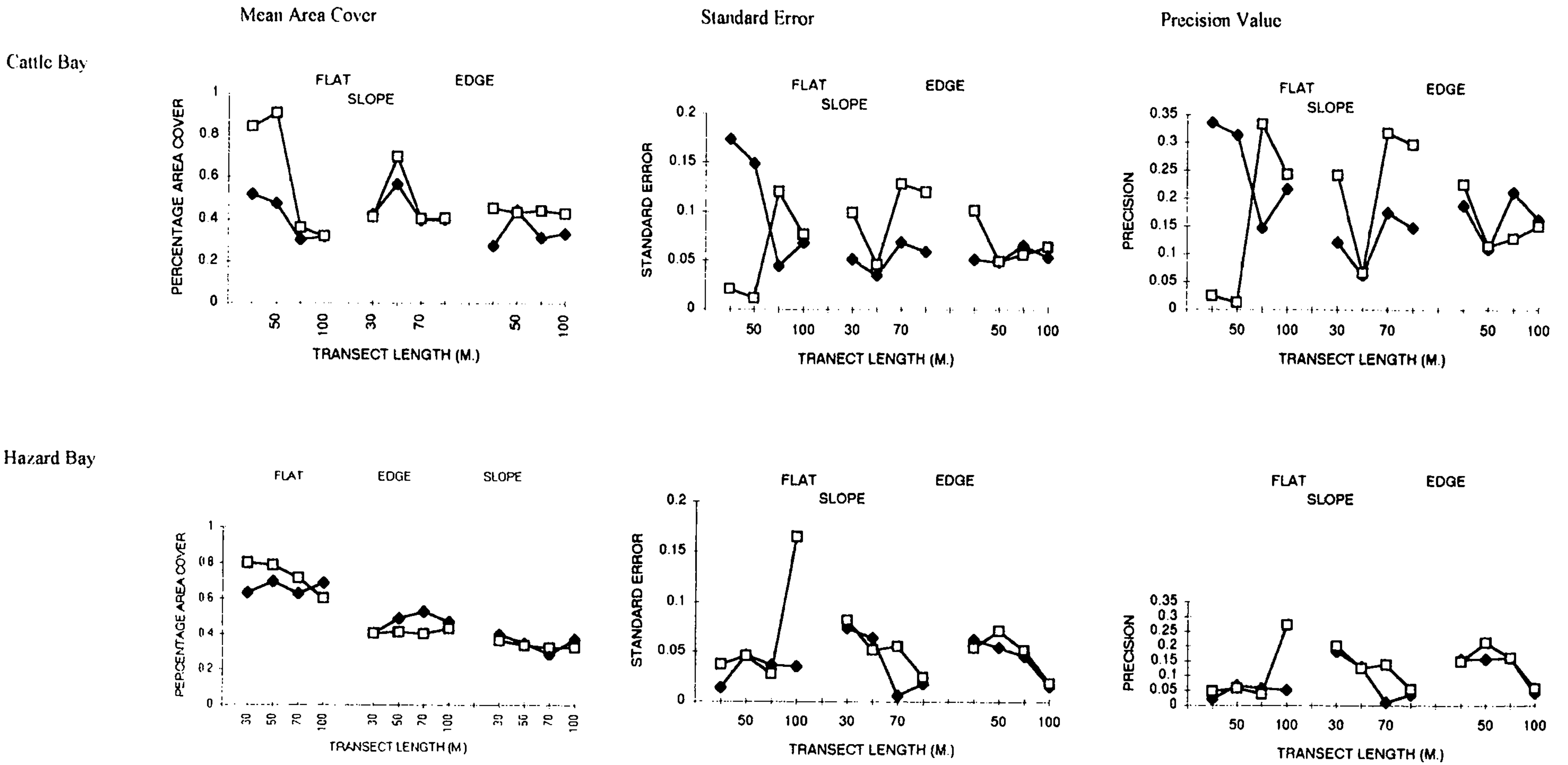
Appendix 3.8. Results of the sampling technique study estimating the Minor component of Living Coral (MLC) at Cattle Bay and Hazard Bay. Graphs show Mean Area Cover, Standard Error and Precision Value; ◆------◆ Fixed Density Point Transect, □------□ Line Intercept Transect.



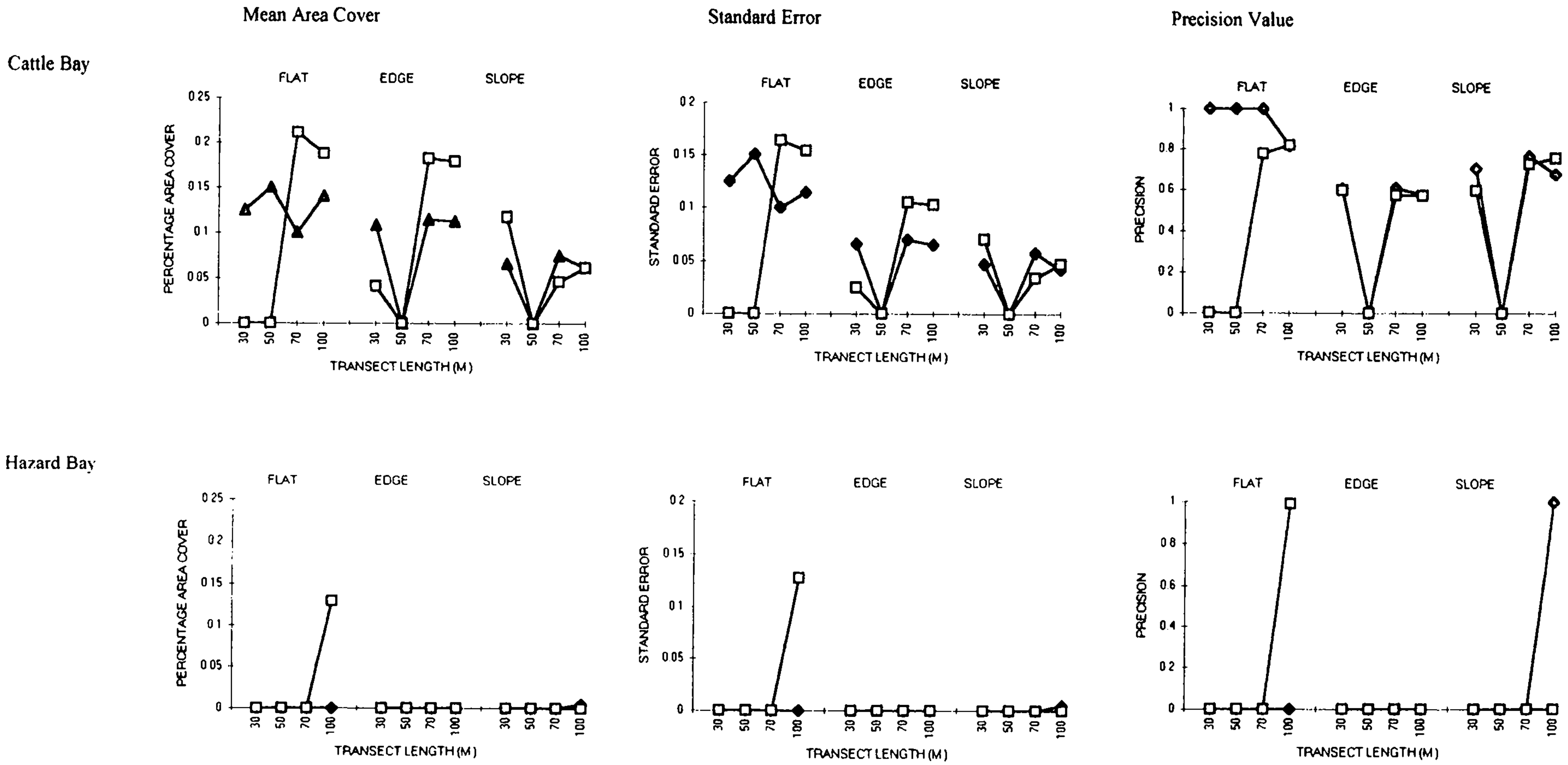
Appendix 3.9. Results of the sampling technique study estimating Fire Coral (FC) at Cattle Bay and Hazard Bay. Graphs show Mean Area Cover, Standard Error, and Precision Value; ◆------◆ Fixed Density Point Transect, □------□ Line Intercept Transect.



Appendix 3.10. Results of the sampling technique study estimating Soft Coral (SC) at Cattle Bay and Hazard Bay. Graphs show Mean Area Cover, Standard Error and Precision Value; ◆----◆ Fixed Density Point Transect, □-----□ Line Intercept Transect.



Appendix 3.11. Results of the sampling technique study estimating Dead Coral (DC) at Cattle Bay and Hazard Bay. Graphs show Mean Area Cover, Standard Error and Precision Value; ◆------◆ Fixed Density Point Transect, ☐------☐ Line Intercept Transect.



Appendix 3.12. Results of the sampling technique study estimating Dead Coral with Algae (DCA) at Cattle Bay and Hazard Bay. Graphs show Mean Area Cover, Standard Error and Precision Value; ◆------◆ Fixed Density Point Transect, □-----□ Line Intercept Transect.

Appendix 4.1. Mean area cover of benthic lifeforms and community parameters of the benthic assemblage from two habitats at eight localities at Phuket, Thailand, during 1994 and 1995. Details of site code name are presented in Fig. 4.8.

| | | Mar-94 | | | | | | | | | | | | | | | | Mar-95 | | | | | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | A1 | a1 | B1 | b1 | C1 | c1 | D1 | d1 | E1 | e1 | F1 | f1 | G1 | g1 | H1 | h1 | A2 | a2 | B2 | b2 | C2 | c2 | D2 | d2 | E2 | e2 | F2 | f2 | G2 | g2 | H2 | h2 | Total cover | Occurrence | Relative occurrence | Habitat width (AH') |
| Dead coral | DC | 0.779 | 0.283 | 0.321 | 0.029 | 0.325 | 0.583 | 0.210 | 0.385 | 0.287 | 0.051 | 0.330 | 0.196 | 0.311 | 0.292 | 0.234 | 0.126 | 0.044 | 0.074 | 0.178 | 0.020 | 0.222 | 0.533 | 0.289 | 0.343 | 0.278 | 0.324 | 0.170 | 0.189 | 0.391 | 0.158 | 0.120 | 0.014 | 0.260 | 32 | 100.00 | 8.80 |
| Massive coral | CM | 0.042 | 0.030 | 0.025 | 0.018 | 0.297 | 0.137 | 0.331 | 0.283 | 0.134 | 0.004 | 0.124 | 0.080 | 0.081 | 0.018 | 0.167 | 0.035 | 0.029 | 0.029 | 0.110 | 0.003 | 0.334 | 0.164 | 0.378 | 0.202 | 0.057 | 0.014 | 0.214 | 0.070 | 0.044 | 0.022 | 0.210 | 0.026 | 0.116 | 32 | 100.00 | 8.25 |
| Coral debris | CD | 0.006 | 0.250 | 0.260 | 0.701 | 0.023 | 0.086 | 0.023 | 0.078 | 0.054 | 0.103 | 0.008 | 0.010 | 0.054 | - | 0.441 | 0.562 | 0.572 | 0.034 | 0.151 | 0.077 | 0.036 | 0.056 | 0.016 | 0.207 | - | 0.094 | 0.061 | - | 0.056 | 0.246 | 0.384 | 0.756 | 0.169 | 29 | 90.63 | 7.58 |
| Encrusting coral | CE | 0.018 | 0.019 | 0.009 | 0.007 | 0.145 | 0.036 | 0.081 | 0.033 | 0.108 | 0.096 | 0.018 | 0.048 | 0.019 | 0.002 | 0.030 | 0.003 | 0.034 | 0.104 | 0.034 | 0.017 | 0.034 | 0.018 | 0.076 | 0.023 | 0.032 | 0.013 | 0.008 | 0.010 | - | 0.010 | - | - | 0.034 | 29 | 90.63 | 8.09 |
| Submassive coral | CS | 0.000 | 0.006 | 0.040 | 0.016 | 0.033 | 0.001 | 0.036 | 0.008 | 0.006 | 0.001 | - | - | 0.024 | 0.002 | 0.009 | 0.003 | 0.002 | - | 0.090 | 0.048 | 0.152 | 0.014 | 0.102 | 0.023 | 0.044 | - | 0.007 | 0.019 | 0.017 | 0.067 | 0.011 | 0.001 | 0.024 | 28 | 87.50 | 7.32 |
| Tabulate Acropora | ACT | 0.060 | 0.039 | 0.006 | 0.006 | 0.006 | 0.019 | - | 0.011 | 0.003 | 0.006 | 0.009 | 0.078 | 0.019 | 0.138 | 0.003 | - | 0.071 | 0.022 | 0.002 | 0.033 | 0.016 | 0.046 | - | 0.004 | 0.011 | 0.016 | 0.013 | 0.040 | 0.030 | 0.214 | - | - | 0.029 | 27 | 84.38 | 7.25 |
| Branching Acropora | ACB | 0.078 | 0.002 | 0.072 | 0.054 | 0.016 | 0.008 | 0.003 | - | 0.012 | 0.011 | 0.311 | 0.242 | 0.417 | 0.526 | - | - | 0.060 | - | 0.029 | 0.072 | 0.004 | - | 0.002 | 0.001 | 0.031 | 0.053 | 0.255 | 0.217 | 0.416 | 0.201 | - | - | 0.097 | 25 | 78.13 | 6.87 |
| Sand | SA | - | - | - | 0.004 | 0.071 | 0.039 | 0.037 | 0.090 | 0.021 | 0.044 | 0.003 | 0.007 | 0.004 | - | 0.096 | 0.252 | 0.023 | 0.014 | 0.002 | - | 0.056 | 0.083 | 0.015 | 0.082 | 0.152 | 0.024 | 0.044 | - | 0.007 | - | 0.258 | 0.185 | 0.050 | 25 | 78.13 | 7.27 |
| Submassive Acropora | ACS | 0.001 | 0.005 | 0.007 | 0.024 | 0.005 | 0.034 | 0.007 | 0.030 | 0.002 | 0.011 | - | - | 0.008 | - | 0.005 | - | 0.001 | - | 0.007 | 0.030 | - | 0.010 | 0.002 | 0.099 | 0.001 | 0.003 | 0.004 | 0.006 | - | 0.002 | 0.006 | 0.001 | 0.010 | 25 | 78.13 | 6.68 |
| Foliose coral | CF | 0.002 | - | 0.009 | 0.090 | 0.016 | 0.009 | 0.037 | 0.002 | 0.268 | 0.661 | 0.148 | 0.311 | 0.053 | 0.019 | - | - | - | - | 0.014 | - | 0.029 | 0.008 | 0.021 | 0.006 | 0.266 | 0.440 | 0.066 | 0.211 | 0.029 | 0.014 | - | - | 0.085 | 24 | 75.00 | 6.43 |
| Branching coral | CB | 0.003 | 0.012 | 0.026 | 0.013 | 0.004 | 0.002 | 0.007 | 0.022 | 0.003 | 0.004 | 0.004 | 0.026 | 0.005 | 0.004 | - | - | - | - | 0.010 | 0.015 | 0.001 | 0.002 | 0.016 | - | 0.018 | 0.007 | - | 0.017 | 0.002 | 0.008 | - | - | 0.007 | 24 | 75.00 | 7.79 |
| Blue coral | BC | - | 0.060 | 0.019 | 0.001 | 0.008 | 0.004 | - | 0.004 | 0.102 | 0.009 | - | - | - | - | 0.001 | 0.002 | - | 0.020 | 0.073 | - | 0.025 | 0.008 | - | 0.002 | 0.089 | 0.011 | - | - | - | - | - | - | 0.014 | 17 | 53.13 | 5.85 |
| Algae on dead coral | DCA | - | 0.231 | - | - | - | - | 0.022 | 0.016 | - | - | 0.047 | 0.003 | - | - | - | - | 0.108 | 0.386 | 0.274 | 0.677 | 0.016 | - | 0.061 | - | - | 0.001 | 0.154 | 0.220 | - | 0.058 | - | - | 0.071 | 15 | 46.88 | 5.71 |
| Soft coral | SC | 0.002 | - | 0.007 | - | - | - | 0.027 | 0.016 | - | - | - | - | 0.004 | - | - | - | - | 0.002 | 0.007 | 0.003 | 0.002 | - | 0.017 | 0.006 | - | - | 0.002 | 0.001 | - | - | - | 0.006 | 0.003 | 14 | 43.75 | 6.09 |
| Fire coral | FC | - | - | - | - | 0.019 | 0.026 | 0.001 | - | - | - | - | - | - | - | 0.015 | 0.015 | - | - | - | - | 0.019 | 0.031 | - | 0.003 | - | - | - | - | 0.002 | - | 0.009 | 0.010 | 0.005 | 11 | 34.38 | 5.80 |
| Corallimorphs | CRLM | - | - | - | - | 0.032 | 0.017 | - | - | - | - | - | - | - | - | - | - | - | - | 0.019 | 0.003 | 0.054 | 0.026 | 0.001 | - | - | - | - | - | - | - | 0.003 | - | 0.005 | 8 | 25.00 | 4.57 |
| Zooanthids | ZOO | - | 0.001 | - | 0.000 | - | - | 0.002 | - | - | - | - | - | - | - | - | - | - | - | - | 0.002 | 0.001 | 0.002 | 0.004 | - | - | - | - | - | - | - | - | - | 0.000 | 7 | 21.88 | 4.73 |
| Sponge | SP | 0.002 | 0.008 | 0.001 | - | - | - | - | - | - | - | - | - | 0.002 | - | - | 0.003 | - | - | - | - | - | - | - | - | - | - | - | - | 0.007 | - | - | 0.002 | 0.001 | 7 | 21.88 | 4.34 |
| Macroalgae | MA | 0.006 | 0.052 | - | 0.036 | - | - | - | - | - | - | - | - | - | - | - | - | 0.055 | 0.315 | - | - | - | - | - | - | - | - | - | - | - | - | - | - | 0.015 | 5 | 15.63 | 2.77 |
| Rock | R | - | - | - | - | - | - | 0.117 | 0.023 | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | 0.005 | 2 | 6.25 | 1.09 |
| | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Total cover | | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | | | | |
| Lifeform richness | | 13 | 14 | 12 | 14 | 14 | 11 | 15 | 14 | 12 | 12 | 10 | 10 | 13 | 8 | 10 | 9 | 11 | 10 | 15 | 13 | 16 | 14 | 14 | 13 | 11 | 12 | 12 | 11 | 11 | 11 | 8 | 9 | | | | |
| Lifeform diversity | | 0.90 | 1.84 | 1.45 | 1.23 | 1.83 | 1.52 | 1.93 | 1.78 | 1.83 | 1.24 | 1.60 | 1.76 | 1.60 | 1.16 | 1.48 | 1.18 | 1.54 | 1.61 | 2.09 | 1.29 | 1.99 | 1.62 | 1.73 | 1.74 | 1.90 | 1.49 | 1.92 | 1.85 | 1.40 | 1.88 | 1.44 | 0.78 | | | | |
| Evenness index | | 0.35 | 0.70 | 0.59 | 0.47 | 0.69 | 0.58 | 0.71 | 0.68 | 0.73 | 0.50 | 0.70 | 0.77 | 0.63 | 0.56 | 0.64 | 0.54 | 0.64 | 0.70 | 0.77 | 0.50 | 0.72 | 0.62 | 0.65 | 0.68 | 0.79 | 0.60 | 0.77 | 0.77 | 0.58 | 0.78 | 0.69 | 0.35 | | | | |

Appendix 4.2. Mean abundance of coral reef fishes and community parameters of each fish family from two habitats at eight localities at Phuket, Thailand, during 1994 and 1995. Details of site code name are presented in Fig. 4.8.

| Species | Abbreviation | Mar 94 |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  | Mar 95 |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  | Total | Relative | Occurrence | Relative | Habitat |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
|  |  | A1 | a1 | B1 | b1 | C1 | c1 | D1 | d1 | E1 | e1 | F1 | f1 | G1 | g1 | H1 | h1 | A2 | a2 | B2 | b2 | C2 | c2 | D2 | d2 | E2 | e2 | F2 | f2 | G2 | g2 | H2 | h2 | abundance | abundance (%) |  | Occurrence (%) | width (AHI) |
| CHAETODONTIDAE |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| *Chaetodon triangulum* | CHTRI | 0.83 | 0.17 | 0.67 | 0.33 | 1.33 | 0.33 | 2.33 | 1.83 | 0.67 | 0.17 | 4.33 | . | 4.00 | 2.67 | 0.17 | . | 0.67 | 1.17 | 1.17 | 1.00 | 1.83 | 2.17 | 2.83 | 2.50 | 1.50 | 0.67 | 3.50 | 2.67 | 2.17 | 2.33 | 0.33 | 0.33 | 292 | 0.32 | 30 | 93.75 | 8.47 |
| *Chaetodon collare* | CHCOL | . | 0.50 |  | . | 0.67 | 1.93 | 0.33 | 0.50 | 4.17 | . | 1.33 | 0.50 | 0.67 | 1.17 | 1.33 | 0.33 | 1.17 | 4.17 | 0.17 |  | 1.00 | 0.17 | 0.17 | . | 2.50 | 1.33 | 0.83 | . | 1.33 | 2.00 | 0.83 | 0.67 | 178 | 0.20 | 25 | 78.13 | 7.87 |
| *Heniochus acuminatus* | HASIN | 0.50 | 0.17 | . |  |  | 0.50 |  | 0.17 | 0.33 | . | 0.67 |  | . | . | 0.33 | 0.33 | 0.17 | . | 0.33 | . | 0.50 | . | 0.50 | 0.17 | . | . | 0.50 | 0.17 | . | 0.17 | 0.17 | . | 34 | 0.04 | 17 | 53.13 | 7.44 |
| *Chaetodon octofasciatus* | CHOCT | . |  | . |  | 1.33 |  |  | . | 0.33 | 0.17 | 2.50 | 1.67 | 0.83 | 1.00 | 0.33 | . |  | . | . | . | 2.50 | . | 0.17 | . | 1.00 | 0.17 | 0.33 | 1.00 | 0.83 | 0.50 | . | . | 100 | 0.11 | 16 | 50.00 | 6.59 |
| *Chaetodon trifasciatus* | CHTRT | 0.33 | 0.67 |  |  | 0.17 | 0.17 |  | 0.17 | 2.33 | . | 0.67 | 2.50 | . | . | 0.50 | . |  | . | 0.33 | 0.33 | . | . | 0.33 | 0.67 | 0.50 | . | . | . | . | 0.17 | . | . | 59 | 0.06 | 15 | 46.88 | 6.22 |
| *Chaetodon plebeius* | CHPLE | 0.17 | 0.17 | . |  |  |  |  | 0.50 | 1.00 | 0.17 | . |  | . | . |  | . | 0.17 | 0.50 | 0.33 | 0.50 | 0.33 |  | . | 0.83 | . | 0.17 | . | . | . |  |  | 0.17 | 30 | 0.03 | 13 | 40.63 | 6.49 |
| *Heniochus pleurotaenia* | HEACU | . |  |  |  | [illegible] | [illegible] |  |  | 0.67 |  | 1.67 | 1.00 | 0.17 |  | 0.17 | . |  | . |  | . |  |  |  |  |  |  | . |  |  |  | 0.17 |  | 45 | 0.05 | 8 | 25.00 | 4.87 |
| *Chaetodon trifascialis* | CHTFL | 0.83 | 0.50 | . |  |  |  |  |  |  | . |  |  |  |  | 0.17 | . | 0.33 | 0.17 | . | . |  | . | 0.17 | . | . | . | . | 0.17 |  | 0.83 |  |  | 19 | 0.02 | 8 | 25.00 | 5.16 |
| *Chaetodon vagabundus* | CHVAG | . | . | . | . | . | . | . | . | . | . | 0.33 | . |  | . |  | . | 0.33 | 0.33 | . | 0.83 | . | 0.33 |  | . | . | . | . | . | . | . |  |  | 13 | 0.01 | 5 | 15.63 | 4.09 |
| *Chaetodon lineolatus* | CHLIN | . | . | . | . | . | 0.17 | . | . | . | . | 0.50 | . |  |  |  | . | . | 0.17 | . | . |  | . |  | . | 0.33 | . | . | 0.17 | . | . | . | . | 8 | 0.01 | 5 | 15.63 | 4.16 |
| *Chaetodon auriga* | CHAUR | . | . | . | . | . | 0.83 |  | . |  |  | . | . | . | . |  | . |  | . | 0.17 | 0.50 |  | 0.50 |  | . | . | . | . | . | . | . | . | . | 12 | 0.01 | 4 | 12.50 | 3.51 |
| *Chaetodon melannotus* | CHMEL | . | . | 0.17 |  |  | . | . | . | . | . | 0.17 | . | 0.50 | 0.33 |  | . |  | . |  | . |  | . |  | . | . | . | . | . | . | . | . | . | 7 | 0.01 | 4 | 12.50 | 3.56 |
| *Chaetodon decussatus* | CHDEC |  |  |  |  |  | . |  |  |  | . |  |  |  |  |  | . |  |  |  | 0.67 | 0.33 | 0.83 |  | . | . | . | . | . | . | . | . | . | 11 | 0.01 | 3 | 9.38 | 2.79 |
| *Heniochus singularius* | HEPLE | . | . | . | . | . | . | . | . | . | . | 0.50 | . |  |  |  | . | . | . | 0.33 | . | 0.33 | . |  | . | . | . | . | . | . | . | . | . | 7 | 0.01 | 3 | 9.38 | 2.90 |
| *Chaetodon lunula* | CHLUN | . | . | . | . | . | 0.17 |  | . |  | . |  |  |  |  |  | . |  | . |  | . |  | . |  | . | . | . | . | . | . | . | . | . | 1 | 0.001 | 1 | 3.13 | 0.00 |
| *Chaetodon rafflesi* | CHRUF | . | . | . | . | . | . |  | . |  | . |  |  |  |  |  | . |  | . |  | . |  | . |  | . | . | . | . | . | . | 0.17 | . | . | 1 | 0.001 | 1 | 3.13 | 0.00 |
| Chaetodontid total abundance |  | 16.00 | 13.00 | 5.00 | 2.00 | 44.00 | 35.00 | 16.00 | 19.00 | 57.00 | 3.00 | 76.00 | 34.00 | 37.00 | 31.00 | 18.00 | 4.00 | 17.00 | 39.00 | 17.00 | 23.00 | 41.00 | 24.00 | 25.00 | 25.00 | 47.00 | 14.00 | 31.00 | 25.00 | 26.00 | 37.00 | 9.00 | 7.00 | 817 | 0.90 |  |  |  |
| Chaetodontid species richness |  | 5 | 6 | 2 | 1 | 5 | 8 | 2 | 5 | 7 | 3 | 10 | 4 | 5 | 4 | 7 | 2 | 6 | 6 | 7 | 6 | 7 | 5 | 6 | 4 | 5 | 4 | 4 | 5 | 3 | 7 | 4 | 3 |  |  |  |  |  |
| Chaetodontid species diversity |  | 1.48 | 1.64 | 0.50 | 0.00 | 1.32 | 1.69 | 0.38 | 1.21 | 1.55 | 1.10 | 1.91 | 1.24 | 1.10 | 1.17 | 1.63 | 0.69 | 1.54 | 1.13 | 1.71 | 1.73 | 1.63 | 1.26 | 1.11 | 1.05 | 1.36 | 1.06 | 0.96 | 1.02 | 1.03 | 1.50 | 1.15 | 0.96 |  |  |  |  |  |
| Chaetodontid evenness |  | 0.92 | 0.91 | 0.73 | 0.00 | 0.82 | 0.81 | 0.54 | 0.75 | 0.80 | 1.00 | 0.83 | 0.90 | 0.68 | 0.85 | 0.84 | 1.00 | 0.86 | 0.63 | 0.88 | 0.97 | 0.84 | 0.78 | 0.62 | 0.76 | 0.84 | 0.77 | 0.69 | 0.63 | 0.93 | 0.77 | 0.83 | 0.87 |  |  |  |  |  |
| Labridae |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| *Thalassoma lunare* | THLUN | 24.67 | 32.33 | 15.83 | 37.17 | 62.17 | 37.17 | 16.17 | 28.33 | 42.67 | 19.17 | 52.17 | 45.00 | 22.67 | 18.50 | 24.33 | 22.17 | 12.33 | 30.17 | 21.83 | 18.83 | 30.50 | 15.33 | 15.50 | 14.00 | 34.83 | 19.67 | 43.50 | 26.67 | 32.83 | 25.00 | 90.83 | 27.17 | 5769 | 6.32 | 32 | 100.00 | 9.10 |
| *Labroides dimidiatus* | LADDI | 8.17 | 4.17 | 5.50 | 5.67 | 4.33 | 9.50 | 2.83 | 2.67 | 6.00 | 2.17 | 3.00 | 2.67 | 3.67 | 3.83 | 1.17 | 2.83 | 3.83 | 2.17 | 3.67 | 2.67 | 1.67 | 3.50 | 2.17 | 1.17 | 4.83 | 3.00 | 2.67 | 3.17 | 2.50 | 2.50 | 1.17 | 2.00 | 665 | 0.73 | 32 | 100.00 | 9.08 |
| *Halichoeres* [illegible] | HAVER | 0.50 | 0.17 | 1.67 | 2.17 | 3.67 | 4.17 | 0.33 | 1.67 | 6.00 | 5.00 | 4.17 | 4.67 | 9.33 | 5.83 | 10.33 | 4.67 | 0.83 | . | 0.83 | 0.33 | 0.67 | 0.50 | 1.50 | 0.67 | 9.83 | 5.00 | 10.50 | 2.93 | 4.67 | 9.50 | 12.17 | 2.17 | 758 | 0.81 | 31 | 96.88 | 8.31 |
| *Halichoeres marginatus* | HAMAR | 3.83 | 7.50 | 2.00 | 11.17 | 0.33 | 7.33 | 1.00 | 4.50 | 4.67 | 2.00 | 2.83 | 0.67 | 1.67 | 1.00 | 0.33 | 2.33 | 3.83 | 1.17 | . | 6.33 | 0.33 | 7.00 | 0.33 | 1.67 | 2.17 | 6.17 | 2.33 | 1.00 | 1.67 | 2.67 | 3.50 | 2.83 | 577 | 0.63 | 31 | 96.88 | 9.46 |
| *Labrichthys unilineatus* | LLUNI | 6.67 | 1.33 | 10.00 | 17.50 | 6.33 | 6.67 | 6.67 | 6.17 | 5.00 | 2.33 | 18.33 | 12.50 | 5.83 | 10.67 | 0.83 | . | 6.17 | 0.50 | 13.67 | 8.63 | 5.33 | 4.00 | 9.33 | 4.50 | 5.67 | 7.33 | 9.50 | 11.00 | 9.00 | 10.33 | 0.17 |  | 1348 | 1.49 | 30 | 93.75 | 8.70 |
| *Halichoeres hortulanus* | HAHOR | 8.83 | 4.33 | 13.67 | 20.33 | 1.00 | 14.50 | 3.83 | 3.00 | 0.33 | 1.83 | 0.33 | 0.50 | 0.17 | 0.17 | 6.67 | 4.83 | 9.00 | 4.33 | . | 10.33 | 2.67 | 4.50 | 2.00 | 2.33 | 1.50 | 2.67 | 0.50 | 1.50 | . | 1.67 | 5.33 | 4.17 | 821 | 0.90 | 30 | 93.75 | 7.97 |
| *Gomphosus caeruleus* | GOCAE | 1.67 | 0.83 | 1.67 | 3.67 | 0.17 | 3.33 | 2.00 | 3.50 | 0.33 | 1.33 | 2.00 | 2.67 | 0.17 | 0.67 | 0.33 | 0.33 | 1.00 | 1.17 | 0.83 | 4.17 | 1.00 | 2.00 | 0.50 | 2.00 | 0.17 | 0.83 | 0.17 | 1.50 |  | 0.33 | 0.33 | . | 244 | 0.27 | 30 | 93.75 | 8.33 |
| *Stethojulis trilineata* | STTRI | 5.00 | 5.33 | 0.17 | 5.00 | 0.33 | 3.67 | 0.67 | 1.83 | 0.67 | 1.17 | 0.17 | . | . | 1.00 | 0.33 | 1.33 | 6.33 | 2.50 | 1.33 | 2.67 | 2.17 | 6.33 | 0.17 | 1.67 | 0.83 | 2.17 | 0.17 | 0.17 | . | 0.67 | 0.33 | 1.67 | 383 | 0.42 | 29 | 90.63 | 7.75 |
| *Halichoeres trimaculatus* | HATRI | 6.83 | 0.17 | 2.17 | 2.33 | 1.33 | 5.50 | 7.17 | 5.00 | 5.67 | 3.67 | 0.33 | . | 0.67 | 0.50 | 1.33 | 0.33 | 4.50 | 0.17 | 0.67 | . | 2.00 | 1.00 | 2.83 | 1.67 | 2.33 | 5.67 | 1.83 | . | 0.33 | . | 0.33 | 1.00 | 416 | 0.46 | 22 | 87.50 | 8.02 |
| *Epibulus insidiator* | EPINS | 3.50 | 3.00 | 1.33 | 1.33 | 1.00 | 1.33 | 0.33 | 0.50 | 0.67 | 0.17 | 3.00 | 3.17 | 1.50 | 3.00 | 0.83 |  | 1.33 | 3.93 | 1.00 | 1.50 | 0.67 | 0.33 |  | 0.17 | 0.17 | 0.50 | 2.83 | 1.67 | 0.67 | 1.17 | . | . | 743 | 0.27 | 23 | 97.50 | 8.29 |
| *Hemigymnus fasciatus* | HOFAS | 0.17 | 0.17 | 0.67 | 1.33 | . | 0.33 | 0.83 | 0.50 | . |  | 0.17 |  | . | 0.17 | 0.33 | 0.33 |  | . | 7.67 |  | 0.50 | . | 0.33 | 0.17 | . | . | 0.50 |  | 0.17 | 0.50 | 0.17 | . | 90 | 0.10 | 19 | 59.38 | 5.48 |
| *Bodianus sp.* | BOSP | 0.33 |  | . | 0.17 | 1.33 | 0.50 | 0.50 | 0.67 | 0.33 | . | 0.33 |  | . | . | . | . | 0.17 | . | . | . | 0.83 | 0.67 | 1.00 | 0.17 | 1.17 | . | 0.50 | . | 0.17 | 0.17 | 0.17 | . | 55 | 0.06 | 18 | 56.25 | 7.28 |
| *Hemigymnus melapterus* | HGMEL | 0.17 | 0.17 | 0.33 | . | 0.17 | 0.50 |  | . | 0.33 | . | 0.67 | . | 0.33 | 0.50 | 0.17 | . | 0.50 | 0.17 | . | 1.00 |  |  |  | . | . | 0.33 | 1.00 | 0.33 | . | 0.33 | . | . | 42 | 0.05 | 17 | 53.13 | 7.26 |
| *Cheilinus trilobatus* | CUTRI | 1.33 | 1.67 | 1.50 | 3.00 | 0.50 | 1.00 | 0.17 | . | . | . | . | . | . | . | . | 0.17 | 2.00 | 2.33 | . | 5.50 | 1.50 | 0.17 |  | . | . | . | . | 0.50 | . | . | 0.33 | 0.50 | 133 | 0.15 | 16 | 50.00 | 6.45 |
| *Cheilinus diagrammus* | CLDIA | . | 0.17 | 0.67 | . |  | . | 0.63 | . | . | . | 1.00 | 1.50 | 0.33 | 0.17 | 0.17 | . | . | . | 0.33 | 0.33 | 0.17 | . | 0.33 | 0.17 | . | . | . | 0.67 | 0.17 | 0.17 |  |  | 43 | 0.05 | 16 | 50.00 | 6.81 |
| *Bodianus mesothorax* | BOMES |  | . | 0.83 | . | 1.17 | 0.67 |  | . | 0.33 | . | 1.33 | 0.83 | 0.33 |  |  | . | . |  | 0.33 | . | 0.67 | 0.50 | 1.17 | . | . | . | 0.50 |  | 0.83 | 0.17 | 0.33 |  | 60 | 0.07 | 15 | 46.88 | 6.79 |
| *Cheilinus chlorourus* | CLCHL |  | 0.33 | 0.17 | 0.33 | 0.33 | 0.17 | . | 0.50 | . | 0.17 | . | 0.83 |  | . | . | . | . | 0.17 | 0.33 | . | 0.33 | 0.17 | 0.17 |  | . | . |  | . | . | 0.33 | . | 0.17 | 27 | 0.03 | 15 | 46.88 | 7.03 |
| *Halichoeres argus* | HAARG | . |  |  | 1.33 |  | 6.67 |  | 1.17 |  | 2.00 | 0.17 | . |  | 1.33 | . | . |  |  |  |  |  | 0.17 |  | 1.00 | . | 17.50 | . | 0.17 | . | 1.50 | 1.17 | 5.83 | 240 | 0.26 | 13 | 40.63 | 4.94 |
| *Cheilinus fasciatus* | CLFAS |  |  |  |  | [illegible] | [illegible] |  |  | 1.00 |  | 8.17 | 4.00 | 1.00 | [illegible] |  |  |  |  |  |  |  |  |  |  | 0.17 | 0.17 | 3.00 | 2.00 | 0.67 | 1.00 |  |  | 140 | 0.15 | 13 | 40.63 | 5.55 |
| [illegible] | [illegible] |  | [illegible] |  | [illegible] |  | [illegible] | [illegible] | [illegible] |  |  |  |  |  |  |  |  | [illegible] | [illegible] |  | 1.50 |  |  |  | [illegible] |  | 0.17 | 0.17 | 2.50 |  |  |  |  | [illegible] | 0.07 | 12 | 37.50 | [illegible] |
| [illegible] | [illegible] | [illegible] |  |  |  |  |  |  |  |  |  |  |  |  |  | [illegible] | 1.17 | 12.50 | [illegible] |  |  |  |  |  | 0.33 |  |  | 0.33 |  |  | 0.33 | 0.17 | 0.51 | 131 | 0.15 | 10 | 31.25 | [illegible] |
| [illegible] | [illegible] | [illegible] | [illegible] |  |  |  |  |  |  |  |  | 0.17 |  |  |  | [illegible] | [illegible] |  |  |  |  |  |  |  | [illegible] |  |  | 0.17 |  |  |  | 0.33 | 4.17 | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] |
| [illegible] | [illegible] |  |  | [illegible] |  | [illegible] |  | [illegible] | [illegible] |  |  |  |  |  |  | [illegible] |  |  |  | [illegible] |  | [illegible] |  | [illegible] |  |  |  | [illegible] |  |  |  |  |  | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] |
| [illegible] | [illegible] | [illegible] | [illegible] |  | [illegible] |  |  | [illegible] | [illegible] |  |  |  |  |  |  |  | [illegible] |  |  |  | [illegible] |  |  |  |  |  | [illegible] |  |  |  |  |  | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] |
| [illegible] | PSEVA |  |  |  |  |  |  |  |  |  |  |  |  |  |  | [illegible] |  |  |  | [illegible] |  |  |  | [illegible] |  |  |  | 1.33 | 0.50 |  |  |  |  | [illegible] | [illegible] | [illegible] | 15.63 | [illegible] |
| [illegible] | [illegible] |  |  |  |  |  |  |  |  | 1.33 |  |  |  |  |  |  |  |  |  |  |  | 0.17 |  |  |  | 1.17 |  | 2.83 |  | 2.67 |  |  |  | 49 | 0.05 | 5 | 15.63 | [illegible] |
| [illegible] | [illegible] |  | 0.17 |  |  |  |  | [illegible] | [illegible] | [illegible] |  |  |  | 0.17 |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  | 13 | 0.01 | 5 | 15.63 | 3.70 |
| [illegible] | [illegible] |  |  |  |  |  |  |  |  |  |  |  |  | 0.17 |  |  |  | 0.50 | 0.17 | 0.67 |  |  |  |  | 0.33 |  |  |  |  |  |  |  |  | 11 | 0.01 | 5 | 15.63 | 4.04 |
| [illegible] | [illegible] |  |  |  | 0.33 |  |  |  |  |  |  |  |  |  |  | 0.67 | 4.67 |  |  |  |  |  |  |  |  |  | 0.50 |  |  |  |  |  |  | 37 | 0.04 | 4 | 12.50 | 2.19 |
| [illegible] | [illegible] |  |  |  |  |  |  |  |  |  |  |  |  |  |  | 3.17 | 0.67 |  |  | 0.33 |  |  |  |  |  |  |  |  |  |  |  | 1.00 |  | [illegible] | 0.03 | 4 | 17.50 | 2.85 |
| [illegible] | LABIC |  |  |  |  |  |  |  |  |  |  | 0.17 | 0.33 |  |  |  |  |  |  |  |  |  |  | 0.17 |  |  |  | 0.33 |  |  |  |  |  | 6 | 0.01 | 4 | 12.50 | 3.71 |
| [illegible] | [illegible] |  |  |  |  |  |  | 2.50 |  |  |  |  |  |  |  |  | 4.50 |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  | 6.33 | 50 | 0.09 | 3 | 9.38 | 2.81 |
| [illegible] | [illegible] |  |  |  |  |  |  |  |  |  |  |  |  |  |  | [illegible] | [illegible] |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  | 0.17 | 7 | 0.01 | 1 | 9.38 | 2.61 |
| [illegible] | [illegible] |  |  |  | [illegible] |  |  |  |  |  |  |  |  |  |  | [illegible] |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  | [illegible] |  | 1 | [illegible] | 1 | [illegible] | [illegible] |
| [illegible] | [illegible] |  |  |  |  |  |  | [illegible] |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  | 0.17 | 0.17 | 4 | 0.004 | 1 | [illegible] | [illegible] |
| [illegible] | [illegible] |  |  |  |  | 1.17 |  |  |  |  |  | 0.17 |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  | 9 | 0.01 | 2 | 6.25 | 1.11 |
| *Halichoeres chloropterus* | HACHL |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  | 0.33 | 0.67 | 6 | 0.01 | 2 | 6.25 | 1.66 |
| [illegible] | [illegible] |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  | 0.81 |  |  |  |  |  |  |  |  | 5 | 0.01 | 1 | 3.13 | 0.00 |
| *Anampses lineatus* | ANLIN | . |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  | 0.33 |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  | 2 | 0.002 | 1 | 3.13 | 0.00 |
| *Anampses melanurus* | ANMEL | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | 0.17 | . | . | . | . | . | . | . | . | . | . | . | . | . | 1 | 0.001 | 1 | 3.13 | 0.00 |
| *Pteragogus cryptus* | PTCRY | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | 0.17 | . | . | . | . | . | 1 | 0.001 | 1 | 3.13 | 0.00 |
| Labridae total abundance |  | 437.00 | 392.00 | 350.00 | 725.00 | 535.00 | 628.00 | 283.00 | 371.00 | 452.00 | 246.00 | 592.00 | 476.00 | 288.00 | 286.00 | 363.00 | 355.00 | 395.00 | 300.00 | 336.00 | 428.00 | 319.00 | 277.00 | 263.00 | 204.00 | 389.00 | 435.00 | 511.00 | 337.00 | 338.00 | 353.00 | 712.00 | 361.00 | 12743 | 13.97 |  |  |  |
| Labridae species richness |  | 17 | 19 | 17 | 19 | 18 | 19 | 20 | 18 | 16 | 12 | 20 | 13 | 15 | 15 | 21 | 17 | 17 | 15 | 17 | 14 | 18 | 15 | 17 | 19 | 13 | 16 | 23 | 16 | 13 | 18 | 20 | 17 |  |  |  |  |  |
| Labridae species diversity |  | 2.12 | 1.85 | 2.03 | 2.11 | 1.33 | 2.20 | 2.16 | 2.01 | 1.65 | 1.84 | 1.64 | 1.55 | 1.67 | 1.85 | 2.06 | 2.14 | 2.31 | 1.54 | 1.85 | 2.16 | 1.74 | 2.04 | 2.05 | 2.15 | 1.59 | 2.10 | 1.84 | 1.82 | 1.45 | 1.87 | 0.99 | 1.96 |  |  |  |  |  |
| Labridae evenness |  | 0.75 | 0.63 | 0.74 | 0.72 | 0.46 | 0.75 | 0.72 | 0.70 | 0.60 | 0.74 | 0.55 | 0.61 | 0.62 | 0.68 | 0.68 | 0.76 | 0.82 | 0.57 | 0.65 | 0.82 | 0.60 | 0.75 | 0.72 | 0.73 | 0.62 | 0.76 | 0.59 | 0.66 | 0.57 | 0.65 | 0.33 | 0.69 |  |  |  |  |  |

## Appendix 4.2. (Continued)

| Species | Abbreviation | Mar-94 A1 | a1 | B1 | b1 | C1 | c1 | D1 | d1 | E1 | e1 | F1 | f1 | G1 | g1 | H1 | h1 | Mar-95 A2 | a2 | B2 | b2 | C2 | c2 | D2 | d2 | E2 | e2 | F2 | f2 | G2 | g2 | H2 | h2 | Total abundance | Relative abundance (%) | Occurrence | Relative Occurrence (%) | Habitat width (AH') |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| POMACENTRIDAE | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| *Pomacentrus moluccensis* | POMOL | 1.67 | 5.00 | 35.00 | 10.83 | 87.50 | 46.67 | 24.17 | 2.67 | 6.67 | 7.50 | 79.17 | 45.83 | 200.00 | 93.33 | 12.50 | 2.50 | 25.00 | 4.17 | 70.00 | 31.67 | 112.50 | 25.00 | 67.50 | 21.67 | 15.83 | 1.33 | 158.33 | 85.00 | 123.33 | 92.17 | 44.17 | 1.50 | 9241 | 10.13 | 32 | 100.00 | 8.07 |
| *Plectroglyphidodon lacrymatus* | PLLAC | 57.50 | 63.33 | 133.33 | 40.00 | 116.67 | 45.00 | 16.67 | 19.00 | 18.33 | 26.67 | 6.83 | 15.00 | 30.00 | 16.67 | . | 0.83 | 17.83 | 96.67 | 40.00 | 95.00 | 11.50 | 37.67 | 21.17 | 15.33 | 3.50 | 7.50 | 2.17 | 24.00 | 0.83 | 24.33 | 0.50 | 0.33 | 6025 | 6.61 | 31 | 96.88 | 7.99 |
| *Pomacentrus adelus* | POADE | . | 5.00 | 4.17 | 8.33 | 22.50 | 28.17 | 4.17 | 1.67 | 3.33 | 25.00 | 5.83 | 10.83 | 30.00 | 15.00 | 150.00 | . | 1.33 | . | 4.17 | 0.83 | 14.17 | 19.17 | 12.50 | 5.83 | 32.17 | 36.83 | 23.33 | 20.00 | 15.50 | 18.67 | 45.83 | 3.00 | 3404 | 3.73 | 29 | 90.63 | 7.59 |
| *Pomacentrus chrysurus* | POCHR | 27.50 | 7.33 | 25.83 | 40.00 | 22.50 | 6.00 | 18.33 | 14.00 | . | 11.67 | . | . | . | 1.67 | 30.00 | 19.17 | 16.67 | 4.50 | 14.17 | 12.50 | 0.50 | 5.33 | 6.50 | 24.50 | 1.17 | 15.83 | . | 0.33 | . | 3.17 | 12.00 | 14.83 | 2136 | 2.34 | 26 | 81.25 | 8.06 |
| *Neoglyphidodon nigroris* | NONIG | 5.17 | 9.83 | 8.67 | . | 10.16 | 10.16 | 7.83 | 0.83 | 10.67 | . | 12.33 | 11.17 | 24.50 | 18.50 | . | . | 0.50 | 1.67 | 10.50 | 0.50 | 10.50 | 1.00 | 14.33 | 1.00 | 6.33 | . | 16.67 | 9.50 | 7.67 | 3.83 | 0.33 | . | 1288 | 1.41 | 26 | 81.25 | 8.03 |
| *Abudefduf sexfasciatus* | ABSEX | 4.33 | . | 16.67 | 13.33 | 5.50 | 5.83 | . | . | 5.00 | 23.33 | 35.00 | 25.83 | 23.33 | 41.67 | 2.50 | . | 2.50 | 55.00 | 21.67 | 5.67 | 23.00 | 3.33 | 1.00 | 0.83 | . | . | 101.66 | 40.00 | 59.17 | 6.67 | 13.33 | . | 3217 | 3.53 | 25 | 78.13 | 7.40 |
| *Neopomacentrus azysron* | NEAZY | . | . | . | . | 198.33 | 60.00 | 143.33 | 400.00 | . | . | 328.33 | 225.00 | . | . | 50.00 | . | 25.00 | 16.67 | 71.67 | 16.67 | 135.83 | 100.00 | 205.00 | 75.00 | 433.33 | 68.33 | 66.67 | 53.33 | 75.00 | 20.00 | 151.67 | 13.33 | 17595 | 19.29 | 23 | 71.88 | 7.49 |
| *Pomacentrus simsiang* | POSIM | . | . | . | . | 9.33 | . | 5.83 | . | 1.00 | . | 0.17 | . | . | . | 13.50 | 37.83 | 4.50 | 0.17 | 2.17 | . | 18.17 | 1.33 | 8.83 | 1.33 | 4.83 | 0.67 | 0.17 | 0.17 | 0.17 | . | . | 15.33 | 753 | 0.83 | 19 | 59.38 | 5.95 |
| *Amblyglyphidodon leucogaster* | AGLEU | 1.67 | . | | . | . | . | 1.67 | . | 1.67 | . | 40.00 | 16.67 | 33.33 | 20.00 | . | . | 2.50 | . | 1.00 | 1.67 | 1.17 | . | 0.83 | 8.50 | 0.33 | . | 176.67 | 36.67 | 60.00 | 8.67 | . | . | 2478 | 2.72 | 18 | 56.25 | 5.15 |
| *Dascyllus trimaculatus* | DATRI | 0.17 | 3.50 | 2.50 | 0.83 | 0.83 | . | 1.00 | 0.83 | . | . | 0.17 | . | . | 0.50 | . | . | 1.00 | . | 5.00 | 3.33 | 2.50 | 0.83 | 1.67 | 0.50 | . | . | . | . | 0.83 | . | . | 0.67 | 160 | 0.18 | 18 | 56.25 | 6.94 |
| *Chromis viridis* | CMVIR | | . | 41.67 | 10.00 | 31.67 | 15.83 | 3.33 | 5.83 | . | . | 73.33 | 78.33 | 171.67 | 250.00 | 35.83 | 4.17 | . | . | . | . | . | . | . | 63.33 | . | . | 196.67 | 33.33 | 318.33 | . | . | . | 8000 | 8.77 | 16 | 50.00 | 5.99 |
| *Amphiprion akallopisos* | AMALK | | 1.17 | . | 0.50 | 0.83 | . | 0.50 | . | . | . | 0.83 | . | 0.33 | 0.50 | 0.83 | 1.67 | . | . | 1.50 | 0.33 | 0.83 | . | 0.33 | . | . | . | 1.00 | . | . | . | 0.33 | 1.67 | 79 | 0.09 | 16 | 50.00 | 7.09 |
| *Chromis ternatensis* | CMTER | . | . | | | . | . | . | . | . | . | . | . | . | . | 100.00 | 35.00 | 6.67 | . | 103.33 | 5.00 | 1.67 | . | 20.83 | 5.00 | 308.33 | 16.67 | 33.33 | 0.83 | 115.00 | . | . | 3.00 | 4528 | 4.95 | 14 | 43.75 | 4.91 |
| *Dascyllus carneus* | DACAR | | . | | | 8.33 | . | 0.83 | . | . | . | 2.50 | 0.17 | | . | 0.50 | 1.67 | . | | 0.50 | . | 0.50 | . | . | 0.50 | 0.17 | . | 4.17 | . | 1.00 | . | 3.67 | 1.83 | 158 | 0.17 | 14 | 43.75 | 5.76 |
| *Neopomacentrus anabatoides* | NEANA | 40.00 | 109.33 | . | 150.00 | . | . | . | . | 156.67 | 50.00 | . | . | 245.00 | 11.67 | | . | . | . | . | . | 146.67 | 1.67 | . | . | . | . | 350.00 | 58.33 | . | . | 1.67 | . | 7920 | 8.69 | 12 | 37.50 | 5.55 |
| *Hemiglyphidodon plagiometopon* | HEPLA | | . | . | 8.33 | | 1.83 | 0.83 | 5.00 | | . | 7.00 | 2.50 | 13.33 | 4.83 | . | . | . | . | . | . | . | . | . | . | . | . | 13.00 | 1.00 | 9.00 | 5.17 | . | . | 431 | 0.47 | 12 | 37.50 | 6.08 |
| *Chrysiptera rollandi* | CSROL | | | | . | . | . | . | | . | . | 18.00 | 6.17 | 10.83 | 1.67 | . | . | . | . | 1.17 | . | 1.50 | . | 1.17 | . | . | . | 34.17 | . | 6.50 | 0.17 | 2.67 | . | 504 | 0.55 | 11 | 34.38 | 4.74 |
| *Neoglyphidodon melas* | NOMEL | 0.83 | 5.00 | | | . | 2.33 | 24.17 | . | 8.33 | 5.00 | . | . | . | . | . | . | . | . | . | . | . | 0.17 | . | 0.17 | . | . | . | 2.00 | . | 0.17 | . | . | 289 | 0.32 | 10 | 31.25 | 4.16 |
| *Dascyllus aruanus* | DAAUR | . | . | | . | . | . | . | . | . | . | . | . | 0.33 | . | 2.67 | 9.67 | . | . | . | . | . | . | . | 0.50 | . | . | . | 0.50 | 0.33 | . | 16.33 | 12.50 | 257 | 0.28 | 8 | 25.00 | 3.84 |
| *Pomacentrus pavo* | POPAV | | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | 6.33 | . | 33.83 | . | 13.83 | . | . | . | 23.33 | . | 15.00 | 3.33 | . | 0.83 | 579 | 0.63 | 7 | 21.88 | 4.38 |
| *Stegastes lividus* | SGLIV | . | . | | 33.33 | . | | | . | . | . | . | . | . | . | . | . | 0.33 | . | 0.83 | 21.17 | . | 1.67 | . | . | . | 10.16 | 0.50 | . | . | . | . | . | 408 | 0.45 | 7 | 21.88 | 3.27 |
| *Pomacentrus alexanderae* | POALE | | . | | | | . | . | . | . | . | . | . | . | . | . | . | 5.00 | . | 13.00 | 3.00 | 2.50 | | 27.33 | . | . | . | 10.50 | 0.33 | . | . | . | . | 370 | 0.41 | 7 | 21.88 | 4.07 |
| *Pomacentrus amboinensis* | POAMB | . | | . | . | . | . | . | . | . | 3.33 | . | . | . | . | 9.50 | 6.67 | . | . | . | . | 0.17 | . | 0.17 | . | . | . | 1.67 | . | . | . | 0.50 | . | 132 | 0.14 | 7 | 21.88 | 3.74 |
| *Plectroglyphidodon dickii* | PLDIC | | 0.17 | . | 1.00 | 0.17 | | . | 2.00 | . | . | . | . | . | . | 0.50 | . | . | . | . | . | . | 1.00 | . | 0.67 | . | . | . | . | . | . | . | . | 57 | 0.06 | 7 | 21.88 | 3.83 |
| *Chromis atripectoralis* | CMART | | . | | | | . | . | . | . | . | . | . | . | . | . | . | 10.00 | | 49.17 | 16.67 | 13.33 | . | . | . | . | . | . | . | . | 147.50 | 13.50 | | 1501 | 1.65 | 6 | 18.75 | 3.41 |
| *Pomacentrus philippinus* | POPHI | . | . | | | | . | . | | | | | | | | | . | . | . | 12.17 | . | . | . | 39.50 | 1.33 | 0.17 | . | 1.50 | . | 1.50 | . | . | . | 337 | 0.37 | 6 | 18.75 | 2.39 |
| *Cheiloprion labiatus* | CLLAB | | . | | . | | . | . | | | | . | 1.17 | 0.67 | 2.00 | | | | . | . | . | . | . | . | . | . | . | 2.00 | 4.67 | | 2.00 | . | . | 75 | 0.08 | 6 | 18.75 | 4.43 |
| *Abudefduf vaigiensis* | ABVAI | . | . | | | . | . | 3.33 | 8.33 | | | . | | . | . | 16.67 | . | . | . | . | . | . | . | . | . | 11.83 | 33.83 | . | . | | | . | . | 444 | 0.49 | 5 | 15.63 | 3.73 |
| *Amphiprion ocellaris* | AMOCE | | 1.17 | 4.33 | | | | . | . | | . | 0.67 | | . | . | . | | . | . | 0.50 | 0.17 | | | . | . | . | . | . | . | . | . | . | . | 41 | 0.04 | 5 | 15.63 | 3.06 |
| *Neopomacentrus cyanomos* | NECYA | | | | | | | . | . | | | . | . | . | . | . | . | . | . | . | . | . | . | . | . | 16.67 | . | 83.33 | | 591.67 | 25.00 | . | . | 4300 | 4.71 | 4 | 12.50 | 1.67 |
| *Chrysiptera unimaculata* | CSUNI | | 2.00 | | | | . | . | 1.33 | | . | | | | . | . | | 0.33 | 1.00 | . | | | . | . | . | . | . | . | . | . | . | . | . | 28 | 0.03 | 4 | 12.50 | 3.34 |
| *Chromis weberi* | CMWEB | | | . | | | | . | . | | | | | | | | . | . | . | 17.50 | . | | . | 39.17 | | . | | | . | | | 40.83 | . | 585 | 0.64 | 3 | 9.38 | 2.82 |
| *Stegastes apicalis* | SGAPI | | | | . | | . | . | . | | | | . | | . | . | . | . | . | . | . | . | 0.33 | . | | . | 0.17 | . | . | . | . | . | 0.33 | 5 | 0.01 | 3 | 9.38 | 2.94 |
| *Chromis cinerascens* | CMCIN | | . | | . | | . | . | . | | | | . | | . | . | . | . | . | . | . | . | . | . | . | . | . | 21.67 | . | . | . | 1.67 | . | 140 | 0.15 | 2 | 6.25 | 0.71 |
| *Chromis flavipectoralis* | CMFLA | . | . | | . | | . | . | . | | | | . | | . | . | | . | . | . | . | . | . | 1.67 | . | . | . | 17.50 | . | . | . | . | . | 115 | 0.13 | 2 | 6.25 | 0.81 |
| *Amphiprion ephippium* | AMEPI | . | . | . | | | | | . | . | | . | . | . | . | . | 1.83 | . | . | . | . | . | . | . | . | . | . | . | 0.83 | . | . | . | . | 16 | 0.02 | 2 | 6.25 | 1.68 |
| *Abudefduf bengalensis* | ABBEN | | . | . | | | | | . | . | | . | . | . | . | . | . | . | . | . | . | . | 0.17 | . | . | . | . | . | . | . | . | . | 0.83 | 6 | 0.01 | 2 | 6.25 | 1.36 |
| *Abudefduf notatus* | ABNOT | . | . | . | | . | | | . | . | | 0.83 | | . | . | | . | . | . | . | . | | . | . | . | . | . | . | . | . | . | 0.17 | . | 6 | 0.01 | 2 | 6.25 | 1.36 |
| *Dischistodus perspicillatus* | DIPER | | | | | 0.33 | | | | | | | | | | | 0.33 | | | | | | | | | | . | | | | | | | 4 | 0.004 | 2 | 6.25 | 1.93 |
| *Amphiprion clarkii* | AMCLA | | | | | | | | | | | | | | | | | 0.17 | | | | | | 0.17 | | | . | | | | | | | 2 | 0.002 | 2 | 6.25 | 1.83 |
| *Chromis margaritifer* | CMMAR | | | | | | | | | | | 3.33 | | | | | | | | | | | | | | | . | | | | | | | 20 | 0.02 | 1 | 3.13 | 0.00 |
| *Pomacentrus brachialis* | POBRA | | | | | | | | | | | | | | | | | | | | | | | | | | | 1.00 | | | | | | 19 | 0.02 | 1 | 3.13 | 0.00 |
| *Stegastes nigricans* | SGNIG | | | | | | | | | | | | | | | | | | | | | | | | | | 0.50 | | | | | | | 1 | 0.001 | 1 | 3.13 | 0.00 |
| *Abudefduf sordidus* | ABSOR | | | | | | | | | | | | | | | | | | | | | | 0.17 | | | | | | | | | | | 1 | 0.001 | 1 | 3.13 | 0.00 |
| Pomacentridae total abundance | | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | 420.00 | [illegible] | [illegible] | | | |
| Pomacentridae species richness | | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | 14 | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | 17 | [illegible] | 11 | [illegible] | 1 | [illegible] | [illegible] | 17 | 14 | | | | | |
| Pomacentridae species diversity | | [illegible] | 1.41 | [illegible] | 1.74 | 1.74 | [illegible] | [illegible] | [illegible] | 1.04 | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | 1.00 | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | 2.00 | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | | | | | |
| Pomacentridae evenness | | [illegible] | [illegible] | [illegible] | 0.70 | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | 0.70 | [illegible] | [illegible] | 0.71 | 0.74 | 0.57 | [illegible] | [illegible] | [illegible] | [illegible] | 0.67 | [illegible] | [illegible] | 0.74 | 0.72 | 0.75 | 0.61 | [illegible] | [illegible] | 0.77 | | | | | |
| Overall abundance | | [illegible] | 405.00 | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | 515.00 | 243.00 | [illegible] | 510.00 | 325.00 | 317.00 | 381.00 | 359.00 | 412.00 | 339.00 | 353.00 | 451.00 | 360.00 | 301.00 | 285.00 | [illegible] | 436.00 | 449.00 | 542.00 | 367.00 | 364.00 | [illegible] | 721.00 | 368.00 | 13560.00 | 14.87 | | | |
| Overall species richness | | 31 | 37 | 26 | 32 | 37 | 37 | 37 | 35 | 32 | 23 | 47 | 29 | 33 | 33 | 42 | 31 | 39 | 29 | 45 | 35 | 44 | 36 | 43 | 40 | 31 | 31 | 52 | 39 | 34 | 40 | 41 | 34 | | | | | |
| Overall species diversity | | 2.352 | 2.101 | 2.15 | 2.436 | 2.156 | [illegible] | 2.163 | 1.214 | 1.912 | 2.339 | 2.097 | 2.041 | 2.035 | 1.947 | 2.303 | 2.558 | 2.927 | 1.892 | 2.638 | 2.535 | 2.278 | 2.181 | 2.33 | 2.333 | 1.493 | 2.438 | 2.546 | 2.557 | 1.924 | 2.273 | 2.196 | 2.722 | | | | | |
| Overall evenness | | 0.639 | 0.592 | 0.645 | 0.703 | 0.597 | 0.742 | 0.599 | 0.341 | 0.552 | 0.746 | 0.545 | 0.612 | 0.582 | 0.557 | 0.616 | 0.745 | 0.799 | 0.562 | 0.693 | 0.713 | 0.602 | 0.609 | 0.619 | 0.632 | 0.435 | 0.724 | 0.644 | 0.579 | 0.546 | 0.616 | 0.591 | 0.772 | | | | | |

Appendix 5.1. Mean area cover of benthic lifeforms from two habitats of 14 localities at Central Great Barrier Reef. Also shown are community parameters; lifeform richness, lifeform diversity, and evenness index, and lifeform parameters; total cover, relative cover, total occurrence, relative occurrence and habitat width of each lifeform. Code names of study sites are represented by a three alphabet-numeric formation. First letter represents areas, M = Magnetic Island, O = Orpheus Island and D = Dunk Island. Second number represents localities in each area which corresponds to number presents in Fig. 5.1. Last letter represents habitat, S = reef slope and E = reef edge.

| Benthic lifeform | Abbreviation | M1S | M1E | M2S | M2E | M3S | M3E | M4S | M4E | O1S | O1E | O2S | O2E | O3S | O3E | O4S | O4E | O5S | O5E | O6S | O6E | O7S | O7E | O8S | O8E | O9S | O9E | DS | Total cover | % of total cover | Site occurrence | % of site occurrence | Habitat width (AH') |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Dead coral | DC | 12.98 | 2.05 | 9.52 | 6.78 | 0.55 | 1.50 | 1.05 | 0.12 | 4.28 | 11.87 | 12.52 | 13.83 | 11.45 | 0.88 | 34.55 | 23.53 | 28.98 | 35.33 | 13.57 | 34.58 | 41.78 | 46.20 | 8.48 | 5.95 | 23.68 | 27.48 | 2.69 | 416.21 | 15.42 | 27 | 100.00 | 7.84 |
| Soft coral | SC | 0.55 | 0.72 | - | 0.38 | 0.50 | - | 1.05 | 1.53 | 37.43 | 52.35 | 45.92 | 15.32 | 34.85 | 53.77 | 1.38 | 46.97 | 3.68 | 21.87 | 1.73 | 22.43 | 2.12 | 16.38 | 8.83 | 34.70 | 14.32 | 22.98 | 0.94 | 442.71 | 16.40 | 25 | 92.59 | 7.22 |
| Branching Acropora | ACB | 4.78 | 1.10 | - | 3.00 | 17.50 | 12.48 | 4.52 | 0.63 | 2.90 | 2.33 | 3.90 | 0.92 | 30.07 | 35.32 | 0.90 | 0.05 | 34.42 | 1.43 | 1.60 | 0.17 | 0.47 | - | 0.42 | 5.08 | 4.62 | 15.55 | 4.06 | 188.21 | 6.97 | 25 | 92.59 | 6.68 |
| Massive coral | CM | 0.50 | - | 1.83 | 0.77 | 0.98 | 1.00 | 1.62 | 1.22 | 0.28 | 1.08 | 1.40 | 4.57 | 0.88 | - | 10.93 | 4.93 | 2.88 | 7.70 | 11.70 | 7.75 | 16.20 | 18.88 | 2.98 | 3.08 | 5.05 | 5.90 | 0.69 | 114.83 | 4.25 | 25 | 92.59 | 7.41 |
| Tabulate Acropora | ACT | 4.10 | 2.93 | 2.18 | 4.27 | 1.82 | 10.17 | 6.43 | 3.88 | 1.03 | 1.93 | 2.80 | 11.62 | - | 2.78 | 1.82 | - | 1.43 | 0.45 | 1.18 | 0.72 | 0.22 | 0.05 | 0.28 | - | - | 0.38 | 7.17 | 69.65 | 2.58 | 23 | 85.19 | 7.30 |
| Coral debris | CD | 2.50 | 1.93 | - | - | 0.72 | 3.88 | 5.17 | 0.55 | 8.97 | 16.10 | 21.57 | 4.00 | - | - | 17.25 | 5.63 | 11.08 | 1.97 | 55.50 | 5.85 | 9.97 | 2.72 | 58.08 | 9.62 | 24.92 | - | 2.06 | 270.02 | 10.00 | 22 | 81.48 | 6.78 |
| Submassive Acropora | ACS | 0.67 | - | - | 0.33 | 0.78 | 0.83 | 0.25 | 0.33 | 0.22 | 0.92 | 0.47 | 18.25 | 0.28 | - | 0.25 | 0.22 | - | 1.17 | 0.17 | 0.30 | - | 0.80 | 0.38 | 6.18 | 0.43 | 0.28 | 0.11 | 33.63 | 1.25 | 22 | 81.48 | 4.80 |
| Algae on dead coral | DCA | 8.50 | 5.83 | 6.83 | 0.93 | 4.68 | 1.55 | 7.30 | 4.05 | 16.07 | - | - | 30.35 | 11.12 | 5.77 | - | - | 0.17 | 7.12 | - | 2.02 | 1.13 | 2.17 | - | 24.62 | 2.93 | 3.32 | 7.61 | 154.08 | 5.71 | 21 | 77.78 | 7.11 |
| Sand | SA | 13.00 | - | 13.22 | 5.02 | 3.38 | 1.50 | 11.68 | - | - | 7.67 | - | - | 2.33 | - | 9.50 | 16.13 | 8.32 | 8.85 | 3.05 | 3.55 | 15.00 | 4.43 | 4.08 | 3.00 | 1.57 | 0.72 | 12.06 | 148.06 | 5.48 | 21 | 77.78 | 7.63 |
| Encrusting coral | CE | 15.33 | 15.17 | 0.53 | 5.22 | 0.90 | 7.33 | 10.85 | 30.93 | 0.55 | - | 0.83 | - | 3.78 | 1.45 | 1.23 | 0.33 | - | - | 2.47 | - | 0.33 | 0.05 | 1.02 | - | 0.88 | 3.83 | 2.19 | 105.23 | 3.90 | 21 | 77.78 | 6.16 |
| Branching coral | CB | 1.55 | - | - | - | 5.68 | 1.00 | - | 0.72 | 0.17 | - | 0.50 | 0.95 | 0.22 | - | 11.05 | 0.33 | 1.08 | 1.82 | 1.55 | 9.32 | 1.53 | 0.20 | 3.57 | 0.75 | 16.55 | 17.72 | 1.89 | 78.14 | 2.89 | 21 | 77.78 | 6.24 |
| Foliose coral | CF | 15.77 | 17.62 | 21.00 | 15.87 | 17.13 | 31.58 | 29.83 | 8.07 | 0.12 | - | - | - | 0.55 | - | 1.50 | 0.22 | - | 0.17 | 3.07 | - | 3.80 | - | 0.72 | - | 0.90 | 0.80 | 2.00 | 170.70 | 6.32 | 19 | 70.37 | 6.22 |
| Submassive coral | CS | - | - | 0.83 | 1.37 | 2.95 | 0.50 | - | 0.67 | 0.42 | 2.47 | - | - | - | - | 2.67 | 1.62 | - | - | 0.22 | 0.80 | 1.55 | 0.67 | - | 0.17 | - | 0.50 | 0.08 | 17.47 | 0.65 | 16 | 59.26 | 6.66 |
| Fire coral | FC | - | - | - | - | - | - | - | - | - | - | - | - | 0.28 | - | 6.98 | - | 7.95 | 10.87 | 4.20 | 7.37 | 5.87 | 7.43 | 4.03 | 2.83 | 4.17 | - | - | 61.98 | 2.30 | 11 | 40.74 | 6.12 |
| Macroalgae | MA | 19.72 | 52.60 | 44.02 | 56.07 | 42.40 | 26.67 | 19.10 | 47.27 | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | 36.56 | 344.39 | 12.76 | 9 | 33.33 | 5.81 |
| Zooanthids | ZOO | - | - | - | - | - | - | 0.53 | - | - | - | - | - | 4.23 | - | - | - | - | 1.22 | - | 5.08 | - | - | 6.47 | 3.75 | - | - | 0.19 | 21.48 | 0.80 | 7 | 25.93 | 4.41 |
| Foraminifera | FOR | - | - | - | - | - | - | - | - | 27.60 | 3.25 | 10.08 | 0.20 | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | 41.13 | 1.52 | 4 | 14.81 | 2.28 |
| Sponge | SP | - | - | - | - | - | - | 0.55 | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | 0.67 | - | - | - | 0.69 | 1.91 | 0.07 | 3 | 11.11 | 2.97 |
| Blue coral | BC | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | 0.25 | - | 0.55 | - | 0.80 | 0.03 | 2 | 7.41 | 1.69 |
| Seagrass | SG | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | 19.00 | 19.00 | 0.70 | 1 | 3.70 | 0.00 |
| Total | | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 2700 | | | | |
| Number of lifeform | | 13 | 9 | 9 | 12 | 14 | 13 | 14 | 13 | 13 | 10 | 10 | 10 | 12 | 6 | 13 | 11 | 10 | 13 | 13 | 13 | 13 | 12 | 14 | 13 | 12 | 13 | 17 | | | | | |
| Lifeform diversity (H') | | 2.16 | 1.44 | 1.59 | 1.52 | 1.79 | 1.88 | 2.06 | 1.45 | 1.62 | 1.54 | 1.56 | 1.85 | 1.68 | 1.07 | 1.97 | 1.44 | 1.72 | 1.89 | 1.56 | 1.93 | 1.78 | 1.57 | 1.58 | 1.93 | 1.97 | 1.86 | 2.01 | | | | | |
| Evenness Index (E) | | 0.81 | 0.66 | 0.72 | 0.61 | 0.68 | 0.73 | 0.78 | 0.57 | 0.63 | 0.67 | 0.68 | 0.81 | 0.68 | 0.60 | 0.77 | 0.60 | 0.75 | 0.74 | 0.61 | 0.75 | 0.70 | 0.63 | 0.60 | 0.75 | 0.79 | 0.73 | 0.71 | | | | | |

Appendix 5.2. Mean abundance (No./150 m$^2$) of wrasse from two habitats of 14 localities at Central Great Barrier Reef. Also shown are community parameters; total abundance, species richness, species diversity, and evenness index, and species parameters; total abundance, relative abundance, total occurrence, relative occurrence and habitat width of each species. Site code name description is the same as in Appendix 5.1.

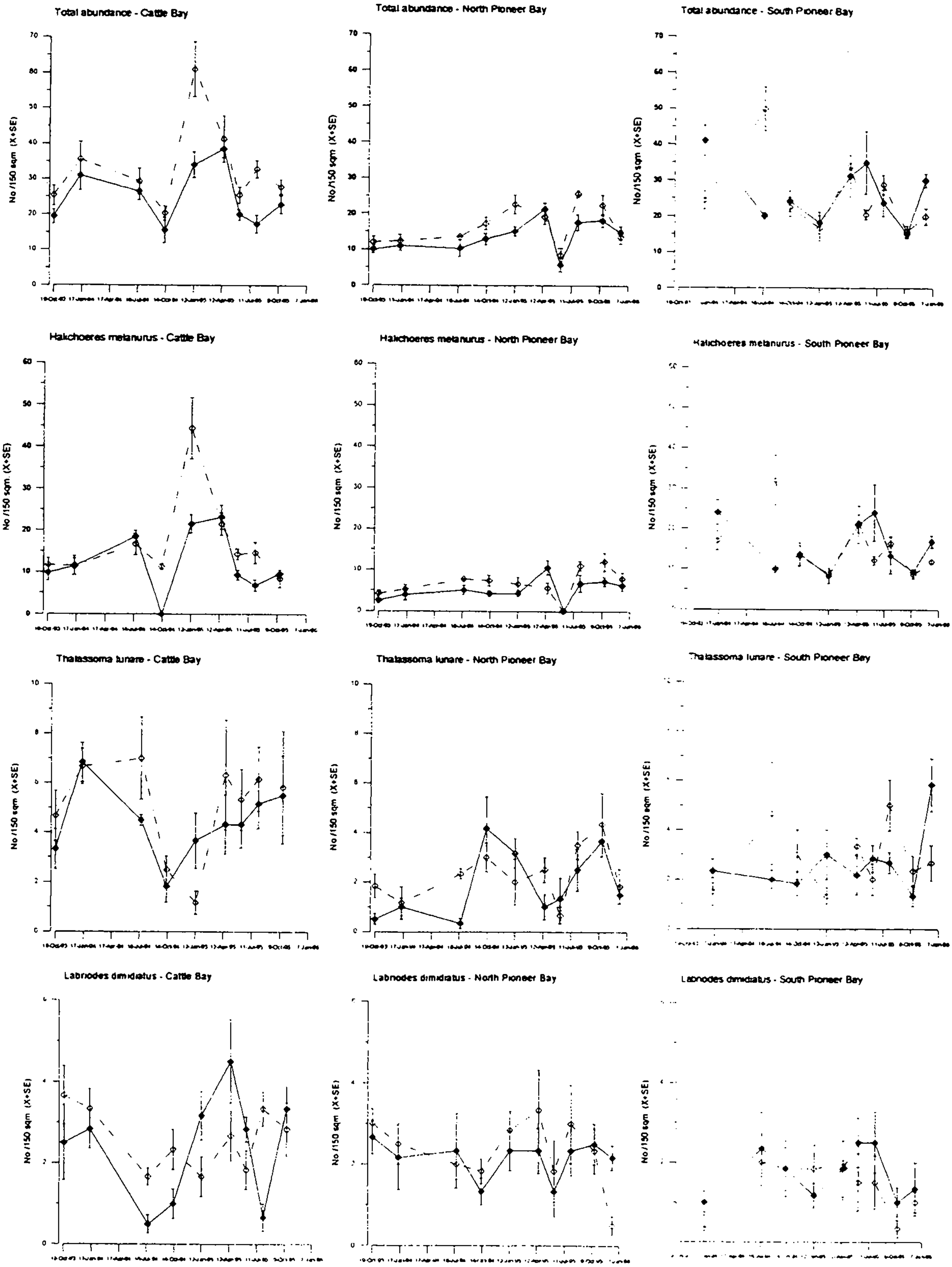
| Species | Abbreviation | M1S | M1E | M2S | M2E | M3S | M3E | M4S | M4E | O1S | O1E | O2S | O2E | O3S | O3E | O4S | O4E | O5S | O5E | O6S | O6E | O7S | O7E | O8S | O8E | O9S | O9E | DS | Total abundance | % of total abundance | Site occurrence | % of site occurrence | Habitat width (AH) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| *Halichoeres melanurus* | HAMEL | 2.33 | 3.67 | 1.00 | 0.33 | 1.83 | 0.17 | 2.50 | 0.17 | 13.33 | 23.00 | 8.00 | . | 4.00 | 1.67 | 3.67 | 10.83 | 1.67 | 8.17 | 4.00 | 7.33 | 2.50 | 4.17 | 9.83 | 11.67 | 4.00 | 8.17 | 1.67 | 838 | 26.52 | 26 | 96.30 | 7.77 |
| *Thalassoma lunare* | THLUN | 0.67 | 0.67 | . | 0.17 | 0.33 | 0.33 | 0.17 | 0.17 | 5.50 | 1.67 | 4.83 | 7.50 | 6.33 | 2.50 | 3.00 | 1.00 | 2.67 | 2.83 | 2.50 | 3.00 | 0.50 | 1.83 | 3.33 | 4.67 | 5.50 | 4.67 | 1.17 | 405 | 12.82 | 26 | 96.30 | 7.81 |
| *Hemigymnus melapterus* | HEMEL | 0.33 | 2.83 | 0.17 | 1.00 | 0.17 | 1.17 | . | 0.17 | 1.67 | 1.00 | 1.00 | 1.83 | 4.67 | 3.83 | 0.67 | 0.67 | 2.00 | 3.50 | 0.67 | 4.00 | 1.17 | 0.50 | 0.50 | . | 1.17 | 2.33 | 1.50 | 231 | 7.31 | 25 | 92.59 | 7.88 |
| *Labroides dimidiatus* | LADIM | 0.67 | . | 0.67 | 0.33 | . | . | 0.17 | . | 2.50 | 3.67 | 3.00 | 3.67 | 5.00 | 1.67 | 1.33 | 0.50 | 1.50 | 2.17 | 2.17 | 2.17 | 2.67 | 3.00 | 2.50 | 3.67 | 1.67 | 3.50 | 2.00 | 301 | 9.53 | 23 | 85.19 | 8.04 |
| *Epibulus insidiator* | EPINS | 0.33 | 0.17 | 0.17 | 0.83 | 0.33 | 0.33 | . | . | 1.17 | 0.50 | 0.50 | . | 1.33 | 1.00 | . | 0.50 | 1.33 | 1.17 | 0.17 | 1.17 | 0.83 | . | 0.17 | 1.17 | 0.83 | 2.33 | . | 98 | 3.10 | 21 | 77.78 | 7.65 |
| *Cheilinus fasciatus* | CLFAS | . | . | . | . | . | 0.33 | . | . | 1.33 | 0.50 | 0.33 | . | 1.33 | 0.17 | 0.83 | 0.17 | 1.33 | 0.33 | 1.00 | 1.00 | 1.17 | 0.33 | 0.67 | . | 1.17 | 0.67 | . | 76 | 2.41 | 17 | 62.96 | 7.24 |
| *Choerodon anchorago* | CRANC | . | . | 0.33 | 0.33 | 0.17 | 0.33 | . | 0.17 | . | . | . | . | 0.33 | 0.50 | 0.17 | 1.00 | 0.33 | 1.17 | 0.67 | 0.50 | 0.33 | . | 0.50 | 1.00 | . | 0.17 | . | 48 | 1.52 | 17 | 62.96 | 7.19 |
| *Cheilinus trilobatus* | CLTRI | 0.33 | 1.50 | 0.17 | 0.17 | 0.83 | 0.83 | 0.17 | . | 0.50 | 0.33 | 0.83 | . | . | . | . | . | . | . | . | . | . | 0.17 | . | 0.17 | 0.33 | 0.33 | 0.33 | 42 | 1.33 | 15 | 55.56 | 6.88 |
| *Stethojulis strigiventer* | STSTR | 0.67 | 0.67 | . | . | . | . | . | . | 0.33 | . | . | . | 1.33 | 0.33 | 0.17 | 1.17 | . | 0.17 | 0.33 | 1.17 | . | 0.50 | 0.50 | 0.67 | . | . | 2.00 | 60 | 1.90 | 14 | 51.85 | 6.55 |
| *Labrichthys unilineatus* | LIUNI | . | . | . | . | 0.33 | 0.17 | . | . | 0.83 | 0.50 | 0.17 | 0.33 | 0.33 | 0.67 | . | . | 0.67 | . | . | 0.17 | . | . | . | 0.17 | 0.17 | 0.50 | . | 30 | 0.95 | 13 | 48.15 | 6.64 |
| *Halichoeres nebulosus* | HANEB | . | 0.17 | 0.17 | 0.67 | . | 0.33 | . | 1.67 | . | . | . | . | . | 0.17 | 0.83 | 3.67 | . | 0.17 | . | . | . | . | 0.50 | 0.33 | . | . | 8.17 | 101 | 3.20 | 12 | 44.44 | 4.44 |
| *Halichoeres dussumieri* | HADUS | 9.67 | 13.50 | 5.33 | 7.00 | 12.50 | 10.50 | 10.50 | 12.17 | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | 3.83 | 510 | 16.14 | 9 | 33.33 | 5.80 |
| *Halichoeres chloropterus* | HACHL | . | . | . | . | . | . | . | . | . | 0.50 | 0.17 | . | . | . | . | 1.00 | . | 1.33 | . | 1.17 | . | 1.17 | 0.67 | 1.00 | 0.17 | . | . | 43 | 1.36 | 9 | 33.33 | 5.54 |
| *Choerodon graphicus* | CRGRA | 0.67 | 0.67 | 0.33 | 0.33 | . | 0.50 | . | 0.33 | . | . | . | . | . | . | . | . | 0.17 | . | . | . | . | . | . | . | . | . | 0.33 | 20 | 0.63 | 8 | 29.63 | 5.43 |
| *Halichoeres biocellatus* | HABIO | 1.00 | 2.67 | 0.17 | . | 0.50 | 1.00 | 0.17 | 1.17 | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | 40 | 1.27 | 7 | 25.93 | 4.41 |
| *Choerodon fasciatus* | CRFAS | . | . | . | . | . | . | . | . | 0.83 | . | 0.17 | . | 0.67 | . | . | . | 0.17 | . | 0.50 | . | 0.67 | . | . | 0.67 | . | . | . | 22 | 0.70 | 7 | 25.93 | 4.95 |
| *Thalassoma hardwicke* | THHAR | . | . | . | . | . | . | . | . | 0.17 | 1.33 | . | 2.17 | . | 0.33 | . | . | . | . | . | 0.33 | . | . | . | . | . | 0.67 | . | 30 | 0.95 | 6 | 22.22 | 3.96 |
| *Choerodon schoenleinii* | CRSCH | . | . | . | . | . | . | . | . | 0.50 | . | . | . | . | . | . | . | . | 0.17 | . | . | 0.17 | . | . | 0.17 | . | . | 2.33 | 20 | 0.63 | 5 | 18.52 | 2.69 |
| *Coris batuensis* | COBAT | . | . | . | . | . | . | . | . | 5.33 | 3.17 | 3.83 | . | 0.33 | . | . | . | . | . | . | . | . | . | . | . | . | . | . | 76 | 2.41 | 4 | 14.81 | 3.17 |
| *Macropharyngodon choati* | MACHO | . | . | . | . | . | . | . | . | 3.17 | 1.83 | 3.67 | . | . | . | . | . | . | . | . | 0.17 | . | . | . | . | . | . | . | 53 | 1.68 | 4 | 14.81 | 3.09 |
| *Halichoeres marginatus* | HAMAR | . | . | . | . | . | . | . | . | . | . | 0.83 | . | . | . | . | . | . | 0.17 | . | . | . | 0.17 | . | . | . | . | 0.17 | 8 | 0.25 | 4 | 14.81 | 2.94 |
| *Cheilinus unifasciatus* | CLUNI | . | . | . | . | . | . | . | . | . | . | . | . | 0.33 | . | . | . | 0.17 | 0.17 | . | . | . | . | . | . | 0.33 | . | . | 6 | 0.19 | 4 | 14.81 | 3.63 |
| *Bodianus axillaris* | BOAXI | . | . | . | . | . | . | . | . | 0.17 | . | . | . | . | . | . | . | 0.17 | . | 0.17 | 0.17 | . | . | . | . | . | . | . | 4 | 0.13 | 4 | 14.81 | 3.77 |
| *Cheilinus undulatus* | CLUND | . | . | . | . | . | . | . | . | . | . | 0.17 | 0.17 | . | . | . | . | . | . | . | . | . | 0.17 | . | . | 0.17 | . | . | 4 | 0.13 | 4 | 14.81 | 3.77 |
| *Hemigymnus fasciatus* | HEFAS | . | . | . | . | . | . | . | . | . | . | 0.33 | . | . | . | . | . | . | . | 0.17 | 0.17 | . | . | . | . | . | . | . | 4 | 0.13 | 3 | 11.11 | 2.84 |
| *Cheilinus oxycephalus* | CLOXY | . | . | . | . | . | . | . | . | . | . | . | . | . | 0.17 | . | . | 0.17 | . | . | . | . | . | . | . | . | 0.17 | . | 3 | 0.09 | 3 | 11.11 | 2.99 |
| *Labropsis australis* | LRAUS | . | . | . | . | . | . | . | . | . | . | . | . | . | 1.83 | . | . | . | . | . | . | . | . | . | . | 0.17 | . | . | 12 | 0.38 | 2 | 7.41 | 0.79 |
| *Cheilinus digrammus* | CLDIA | . | . | . | . | . | . | . | . | 0.83 | 0.33 | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | 7 | 0.22 | 2 | 7.41 | 1.62 |
| *Gomphosus varius* | GOVAR | . | . | . | . | . | . | . | . | . | . | . | 0.17 | . | . | . | . | . | . | . | 0.17 | . | . | . | . | . | . | . | 2 | 0.06 | 2 | 7.41 | 1.88 |
| *Cirrhilabrus punctatus* | CIPUN | . | . | . | . | . | . | . | . | 6.67 | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | 40 | 1.27 | 1 | 3.70 | 0.00 |
| *Thalassoma jansenii* | THJAN |  |  |  |  |  |  |  |  |  |  | . | 3.33 |  |  |  |  |  |  |  |  |  |  |  |  |  |  | . | 20 | 0.63 | 1 | 3.70 | 0.00 |
| *Bodianus mesothorax* | BOMES |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  | 0.33 |  | . | 2 | 0.06 | 1 | 3.70 | 0.00 |
| *Cheilio inermis* | CHINN | . | . | . | . | . | . | . | . | 0.17 | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | 1 | 0.03 | 1 | 3.70 | 0.00 |
| *Cheilinus chlorourus* | CLCHL | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | 0.17 | . | . | . | . | 1 | 0.03 | 1 | 3.70 | 0.00 |
| *Coris gaimard* | COGAI | . | . | . | . | . | . | . | . | . | . | 0.17 | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | 1 | 0.03 | 1 | 3.70 | 0.00 |
| *Macropharyngodon negrosensis* | MANEG | . | . | . | . | . | . | . | . | . | . | 0.17 | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | . | 1 | 0.03 | 1 | 3.70 | 0.00 |
| Total abundance |  | 100 | 159 | 51 | 67 | 102 | 96 | 82 | 96 | 270 | 230 | 169 | 115 | 156 | 89 | 64 | 123 | 74 | 129 | 74 | 136 | 60 | 72 | 116 | 152 | 96 | 141 | 141 | 3160 |  |  |  |  |
| Species richness |  | 10 | 10 | 10 | 10 | 9 | 12 | 6 | 8 | 18 | 13 | 17 | 8 | 12 | 13 | 8 | 10 | 13 | 13 | 11 | 15 | 9 | 10 | 11 | 12 | 13 | 11 | 11 |  |  |  |  |  |
| Species diversity (H') |  | 1.51 | 1.59 | 1.39 | 1.42 | 1.04 | 1.40 | 0.73 | 0.91 | 2.24 | 1.54 | 2.12 | 1.61 | 2.03 | 2.16 | 1.69 | 1.58 | 2.18 | 1.91 | 1.92 | 2.09 | 1.92 | 1.77 | 1.61 | 1.69 | 1.92 | 1.85 | 1.98 |  |  |  |  |  |
| Evenness Index (E) |  | 0.66 | 0.69 | 0.60 | 0.61 | 0.47 | 0.56 | 0.41 | 0.44 | 0.78 | 0.60 | 0.75 | 0.78 | 0.82 | 0.84 | 0.81 | 0.69 | 0.85 | 0.74 | 0.80 | 0.77 | 0.87 | 0.77 | 0.67 | 0.68 | 0.75 | 0.77 | 0.83 |  |  |  |  |  |

Appendix 5.3. Mean abundance and standard error (SE) of wrasse at three localities of Orpheus Island over 10 sampling times. Also shown are community parameters: mean total abundance, species richness, species diversity, and evenness index, and species parameters: total abundance, relative abundance, total occurrence, relative abundance and habitat width of each species.

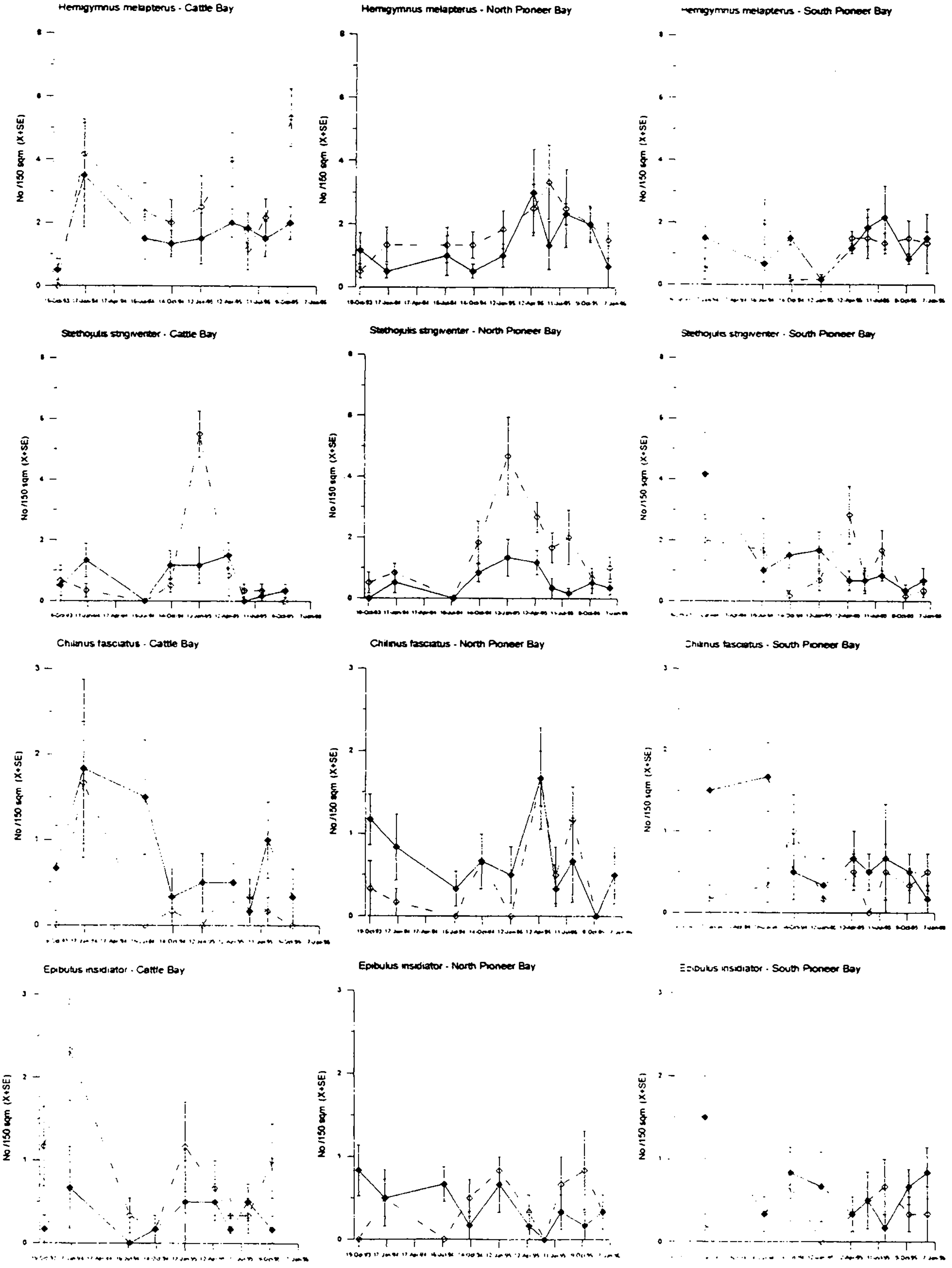
| | | Cattle Bay | | | | North Pioneer Bay | | | | South Pioneer Bay | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Species | Abbreviation | Slope Mean | SE | Edge Mean | SE | Slope Mean | SE | Edge Mean | SE | Slope Mean | SE | Edge Mean | SE | Total abundance | % of total abundance | Site occurrence | % of site occurrence | Habitat width (AH') |
| *Thalassoma lunare* | THLUN | 4.39 | 0.47 | 5.07 | 0.66 | 1.92 | 0.43 | 2.32 | 0.34 | 2.67 | 0.43 | 2.87 | 0.43 | 1064.00 | 13.64 | 56.00 | 100.00 | 10.51 |
| *Labroides dimidiatus* | LADIM | 2.37 | 0.45 | 2.59 | 0.25 | 2.15 | 0.14 | 2.32 | 0.26 | 1.72 | 0.21 | 1.37 | 0.22 | 703.00 | 9.01 | 56.00 | 100.00 | 10.67 |
| *Hemigymnus melapterus* | HEMEL | 1.74 | 0.27 | 2.63 | 0.54 | 1.35 | 0.26 | 1.82 | 0.25 | 1.26 | 0.21 | 1.11 | 0.22 | 554.00 | 7.10 | 55.00 | 98.21 | 10.42 |
| *Halichoeres melanurus* | HAMEL | 12.24 | 2.49 | 17.17 | 3.62 | 5.03 | 0.89 | 6.72 | 1.08 | 15.61 | 2.06 | 15.70 | 2.44 | 3984.00 | 51.06 | 54.00 | 96.43 | 10.36 |
| *Epibulus insidiator* | EPINS | 0.31 | 0.08 | 0.83 | 0.23 | 0.38 | 0.09 | 0.40 | 0.10 | 0.65 | 0.13 | 0.37 | 0.07 | 164.00 | 2.10 | 50.00 | 89.29 | 10.07 |
| *Stethojulis strigiventer* | STSTR | 0.69 | 0.20 | 0.94 | 0.58 | 0.52 | 0.15 | 1.58 | 0.43 | 1.28 | 0.39 | 1.13 | 0.32 | 344.00 | 4.41 | 49.00 | 87.50 | 9.61 |
| *Cheilinus fasciatus* | CLFAS | 0.76 | 0.19 | 0.31 | 0.18 | 0.67 | 0.15 | 0.50 | 0.17 | 0.72 | 0.17 | 0.39 | 0.10 | 188.00 | 2.41 | 47.00 | 83.93 | 7.13 |
| *Choerodon anchorago* | CRANC | 0.44 | 0.10 | 0.74 | 0.22 | 0.22 | 0.07 | 0.22 | 0.09 | 0.35 | 0.10 | 0.35 | 0.11 | 128.00 | 1.64 | 42.00 | 75.00 | 9.52 |
| *Halichoeres chloropterus* | HACHL | 0.26 | 0.07 | 0.56 | 0.12 | 0.00 | 0.00 | 0.23 | 0.12 | 0.35 | 0.08 | 0.35 | 0.09 | 96.00 | 1.23 | 36.00 | 64.29 | 9.28 |
| *Choerodon fasciatus* | CRFAS | 0.13 | 0.08 | 0.07 | 0.07 | 1.17 | 0.20 | 0.05 | 0.03 | 0.33 | 0.09 | 0.11 | 0.08 | 108.00 | 1.38 | 26.00 | 46.43 | 8.11 |
| *Coris batuensis* | COBAT | 0.09 | 0.04 | 0.13 | 0.09 | 0.00 | 0.00 | 0.10 | 0.07 | 0.28 | 0.16 | 0.39 | 0.12 | 54.00 | 0.69 | 23.00 | 41.07 | 7.65 |
| *Labrichthys unilineatus* | LIUNI | 0.09 | 0.06 | 0.74 | 0.16 | 0.00 | 0.00 | 0.02 | 0.02 | 0.24 | 0.16 | 0.15 | 0.06 | 67.00 | 0.86 | 21.00 | 37.50 | 7.48 |
| *Cheilinus digrammus* | CLDIA | 0.13 | 0.05 | 0.17 | 0.06 | 0.05 | 0.04 | 0.00 | 0.00 | 0.06 | 0.04 | 0.28 | 0.18 | 37.00 | 0.47 | 18.00 | 32.14 | 7.07 |
| *Choerodon schoenleinii* | CRSCH | 0.17 | 0.06 | 0.11 | 0.07 | 0.05 | 0.03 | 0.05 | 0.04 | 0.04 | 0.02 | 0.06 | 0.04 | 26.00 | 0.33 | 18.00 | 32.14 | 7.51 |
| *Cheilinus trilobatus* | CLTRI | 0.04 | 0.04 | 0.20 | 0.11 | 0.00 | 0.00 | 0.02 | 0.02 | 0.11 | 0.05 | 0.07 | 0.04 | 24.00 | 0.31 | 14.00 | 25.00 | 6.65 |
| *Cheilinus chlorourus* | CLCHL | 0.04 | 0.02 | 0.07 | 0.04 | 0.00 | 0.00 | 0.05 | 0.04 | 0.06 | 0.04 | 0.17 | 0.08 | 21.00 | 0.27 | 13.00 | 23.21 | 6.67 |
| *Thalassoma janseni* | THJAN | 0.00 | 0.00 | 0.19 | 0.07 | 0.00 | 0.00 | 0.07 | 0.04 | 0.07 | 0.07 | 0.11 | 0.06 | 24.00 | 0.31 | 12.00 | 21.43 | 6.43 |
| *Bodianus axillaris* | BOAXI | 0.00 | 0.00 | 0.06 | 0.04 | 0.05 | 0.04 | 0.05 | 0.04 | 0.20 | 0.11 | 0.02 | 0.02 | 21.00 | 0.27 | 12.00 | 21.43 | 6.18 |
| *Thalassoma hardwicke* | THHAR | 0.02 | 0.02 | 0.48 | 0.19 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.09 | 0.04 | 32.00 | 0.41 | 10.00 | 17.86 | 5.38 |
| *Bodianus mesothorax* | BOMES | 0.00 | 0.00 | 0.02 | 0.02 | 0.08 | 0.04 | 0.03 | 0.03 | 0.07 | 0.04 | 0.04 | 0.02 | 14.00 | 0.18 | 10.00 | 17.86 | 6.11 |
| *Hemigymnus fasciatus* | HEFAS | 0.00 | 0.00 | 0.04 | 0.04 | 0.05 | 0.05 | 0.15 | 0.15 | 0.19 | 0.17 | 0.17 | 0.07 | 33.00 | 0.42 | 9.00 | 16.07 | 5.21 |
| *Cheilinus undulatus* | CLUND | 0.11 | 0.06 | 0.00 | 0.00 | 0.03 | 0.03 | 0.05 | 0.04 | 0.02 | 0.02 | 0.02 | 0.02 | 13.00 | 0.17 | 8.00 | 14.29 | 5.43 |
| *Halichoeres marginatus* | HAMAR | 0.87 | 0.87 | 0.04 | 0.02 | 0.00 | 0.00 | 0.02 | 0.02 | 0.00 | 0.00 | 0.07 | 0.04 | 54.00 | 0.69 | 7.00 | 12.50 | 1.68 |
| *Halichoeres nebulosus* | HANEB | 0.06 | 0.06 | 0.04 | 0.04 | 0.00 | 0.00 | 0.00 | 0.00 | 0.11 | 0.09 | 0.43 | 0.28 | 34.00 | 0.44 | 6.00 | 10.71 | 4.06 |
| *Macropharyngodon choati* | MACHO | 0.02 | 0.02 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.11 | 0.08 | 0.00 | 0.00 | 7.00 | 0.09 | 3.00 | 5.36 | 2.60 |
| *Choerodon graphicus* | CRGRA | 0.02 | 0.02 | 0.02 | 0.02 | 0.00 | 0.00 | 0.00 | 0.00 | 0.02 | 0.02 | 0.00 | 0.00 | 3.00 | 0.04 | 3.00 | 5.36 | 2.99 |
| *Halichoeres dussumeri* | HADUS | 0.02 | 0.02 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.04 | 0.04 | 3.00 | 0.04 | 2.00 | 3.57 | 1.74 |
| *Halichoeres biocellatus* | HABIO | 0.00 | 0.00 | 0.00 | 0.00 | 0.02 | 0.02 | 0.00 | 0.00 | 0.02 | 0.02 | 0.00 | 0.00 | 2.00 | 0.03 | 2.00 | 3.57 | 1.88 |
| Mean total abundance* | | 150.00 | 16.01 | 199.33 | 24.24 | 82.40 | 8.77 | 100.50 | 10.64 | 159.22 | 16.91 | 155.11 | 21.36 | 7802.00 | 100.00 | | | |
| Species richness | | 11.67 | 1.04 | 13.11 | 0.79 | 9.50 | 0.45 | 9.50 | 0.78 | 13.67 | 0.73 | 14.00 | 0.37 | | | | | |
| Species diversity (H') | | 1.55 | 0.09 | 1.63 | 0.10 | 1.80 | 0.05 | 1.68 | 0.08 | 1.57 | 0.05 | 1.52 | 0.04 | | | | | |
| Evenness Index (E) | | 0.64 | 0.02 | 0.63 | 0.03 | 0.80 | 0.02 | 0.76 | 0.02 | 0.61 | 0.03 | 0.58 | 0.02 | | | | | |

* average across nine or ten times

Appendix 5.4. Temporal patterns of some dominance species of wrasse from two habitats of three localities at Orpheus Island, ◆ = reef slope, □ = reef edge.

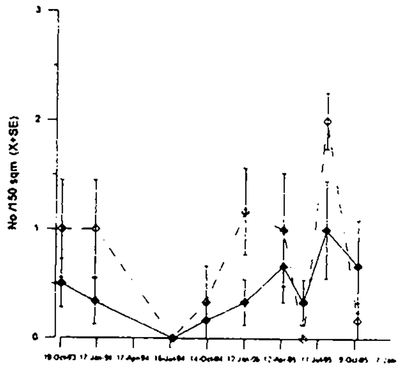


## Appendix 5.4. (Continued).

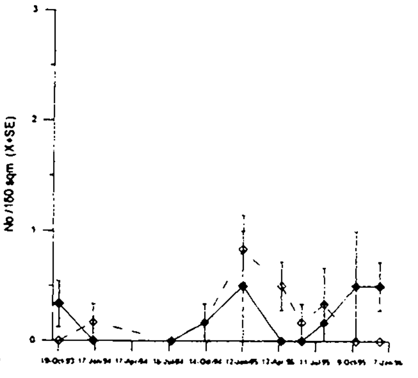


## Appendix 5.4. (Continued)

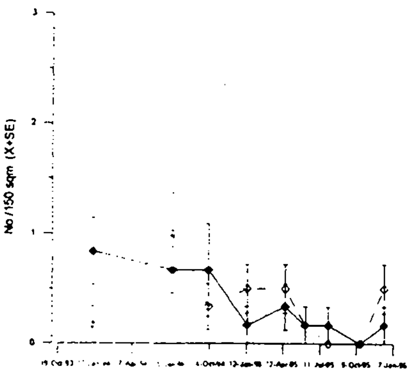
Choerodon ancherago - Cattle Bay



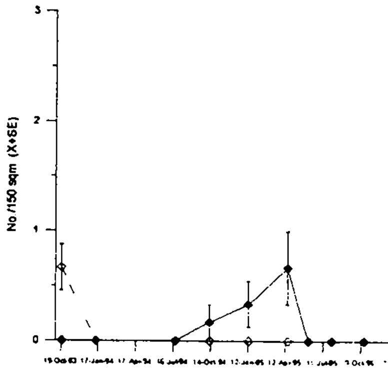
Choerodon ancherago - North Pioneer Bay



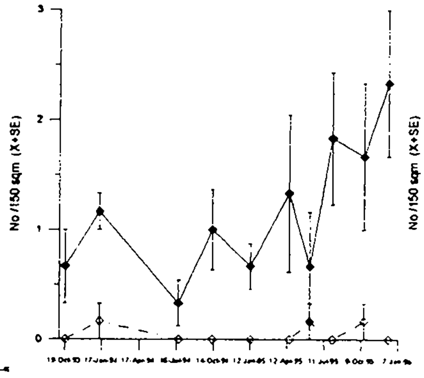
Choerodon ancherago - South Pioneer Bay



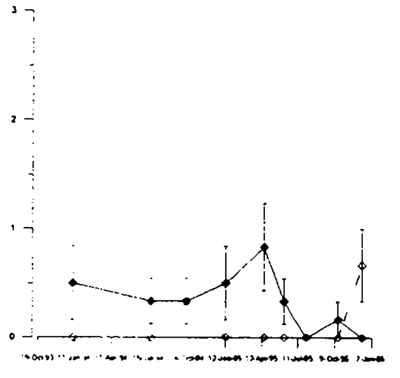
Choerodon fasciatus - Cattle Bay



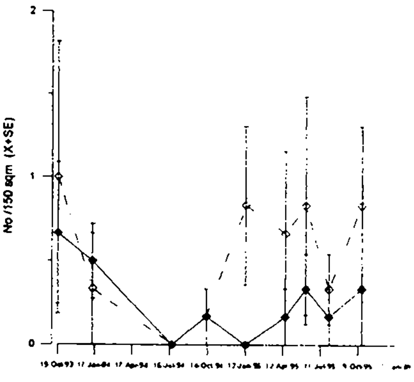
Choerodon fasciatus - North Pioneer Bay



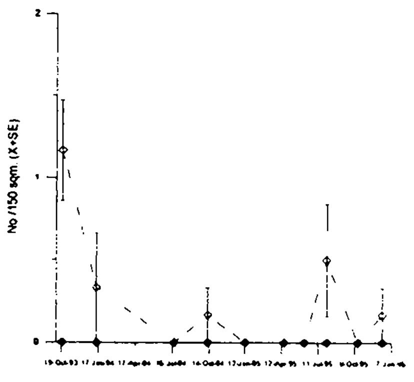
Choerodon fasciatus - South Pioneer Bay



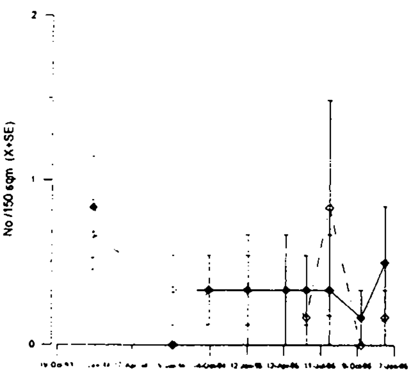
Halichoeres chloropterus - Cattle Bay



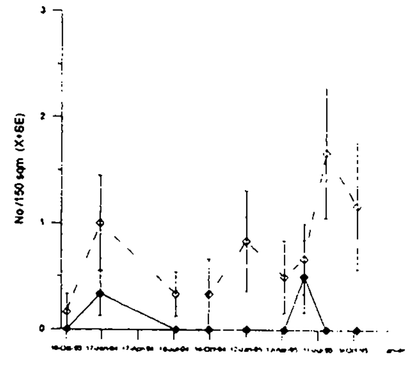
Halichoeres chloropterus - North Pioneer Bay



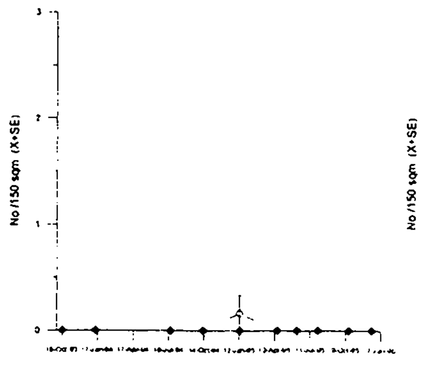
Halichoeres chloropterus - South Pioneer Bay



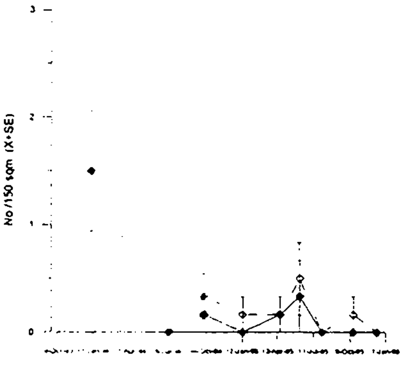
Labichthys unilineatus - Cattle Bay



Labichthys unilineatus - North Pioneer Bay



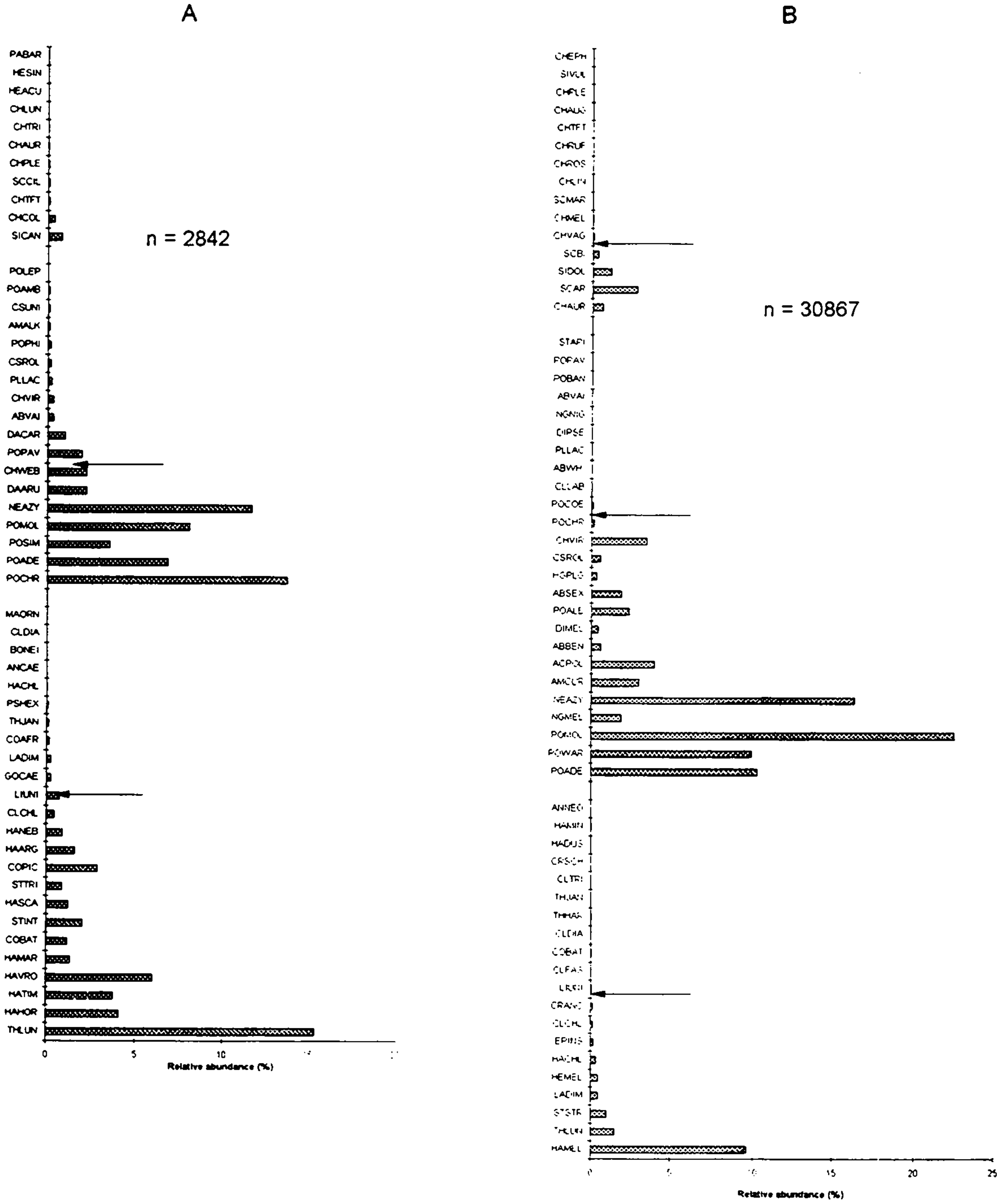
Labichthys unilineatus - South Pioneer Bay



Appendix 6.1. List of coral reef fishes studied during the habitat degradation experiment at Maiton Island (Phuket, Thailand) and Orpheus Island (central Great Barrier Reef, Australia).

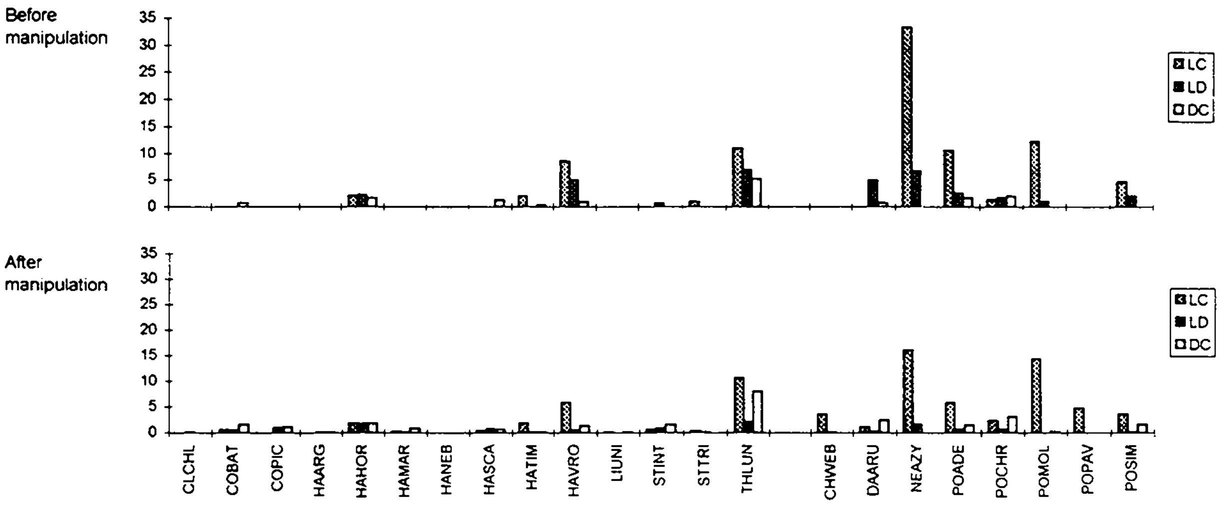
| Species | abbreviation | Maiton Island | Orpheus Island | Species | abbreviation | Maiton Island | Orpheus Island |
|---|---|---|---|---|---|---|---|
| LABRIDAE | | | | *Chrysiptera rollandi* | CSROL | * | * |
| *Anampses caeruleopunctatus* | ANCAE | * | | *Chrysiptera unimaculata* | CSUNI | * | |
| *Anampses neoguinaicus* | ANNEO | | * | *Dascyllus aruanus* | DAARU | * | |
| *Bodianus neilli* | BONEI | * | | *Dascyllus carneus* | DACAR | * | |
| *Cheilinus chlorourus* | CLCHL | * | * | *Dischistodus melanotus* | DIMEL | | * |
| *Cheilinus digrammus* | CLDIA | * | * | *Dischistodus pseudochrysopoecilus* | DIPSE | | * |
| *Cheilinus fasciatus* | CLFAS | | * | *Hemiglyphidodon plagiometopon* | HGPLG | | * |
| *Cheilinus trilobatus* | CLTRI | | * | *Neopomacentrus azysron* | NEAZY | * | * |
| *Coris africana* | COAFR | * | | *Neoglyphidodon nigroris* | NGNIG | | * |
| *Coris batuensis* | COBAT | * | * | *Neoglyphidodon melas* | NGMEL | | * |
| *Choerodon anchorago* | CRANC | | * | *Plectroglyphidodon lacrymatus* | PLLAC | * | * |
| *Choerodon schoenleinii* | CRSCH | | * | *Pomacentrus adelus* | POADE | * | * |
| *Coris pictoides* | COPIC | * | | *Pomacentrus alexanderae* | POALE | | * |
| *Epibulus insidiator* | EPINS | | * | *Pomacentrus amboinensis* | POAMB | * | |
| *Gomphosus caeruleus* | GOCAE | * | | *Pomacentrus bankanensis* | POBAN | | * |
| *Halichoeres argus* | HAARG | * | | *Pomacentrus coelestis* | POCOE | | * |
| *Halichoeres chloropterus* | HACHL | * | * | *Pomacentrus chrysurus* | POCHR | * | * |
| *Halichoeres dussumeri* | HADUS | | * | *Pomacentrus moluccensis* | POMOL | * | * |
| *Halichoeres hortulanus* | HAHOR | * | | *Pomacentrus pavo* | POPAV | * | * |
| *Halichoeres marginatus* | HAMAR | * | | *Pomacentrus philippinus* | POPHI | * | |
| *Halichoeres melanurus* | HAMEL | | * | *Pomacentrus similis* | POSIM | * | |
| *Halichoeres miniatus* | HAMIN | | * | *Pomacentrus wardi* | POWAR | | * |
| *Halichoeres nebulosus* | HANEB | * | | *Pomacentrus lepidogenys* | POLEP | * | |
| *Halichoeres scapularis* | HASCA | * | | *Stegastes apicalis* | STAPI | | * |
| *Halichoeres timorensis* | HATIM | * | | | | | |
| *Halichoeres vrolikii* | HAVRO | * | | OTHER MAJOR SPECIES | | | |
| *Hemigymnus melapterus* | HEMEL | | * | *Chaetodon auriga* | CHAUG | | * |
| *Labroides dimidiatus* | LADIM | * | * | *Chaetodon aureofasciatus* | CHAUR | * | * |
| *Labrichthys unilineatus* | LIUNI | * | * | *Chaetodon collare* | CHCOL | * | |
| *Macropharyngodon ornatus* | MAORN | * | | *Chaetodon lunula* | CHLUN | * | |
| *Pseudocheilinus hexataenia* | PSHEX | * | | *Chaetodon melannotus* | CHMEL | | * |
| *Stethojulis interrupta* | STINT | * | | *Chaetodon ephippium* | CHEPH | | * |
| *Stethojulis trilineata* | STTRI | * | | *Chaetodon lineolatus* | CHLIN | | * |
| *Stethojulis strigiventer* | STSTR | | * | *Chaetodon plebeius* | CHPLE | * | * |
| *Thalassoma hardwicke* | THHAR | | * | *Chaetodon rafflesi* | CHRAF | | * |
| *Thalassoma jansenii* | THJAN | * | * | *Chaetodon trifasciatus* | CHTFT | * | * |
| *Thalassoma lunare* | THLUN | * | * | *Chaetodon triangulum* | CHTRI | * | |
| | | | | *Chaetodon vagabandus* | CHVAG | | * |
| POMACENTRIDAE | | | | *Chelmon rostratus* | CHROS | | * |
| *Abudefduf bengalensis* | ABBEN | | * | *Heniochus acuminatus* | HEACU | * | |
| *Abudefduf sexfasciatus* | ABSEX | | * | *Heniochus singularius* | HESIN | * | |
| *Abudefduf vaigiensis* | ABVAI | * | * | *Scarus juvenile* | SCAR | | * |
| *Abudefduf whitleyi* | ABWHI | | * | *Siganus canaliculatus* | SICAN | * | |
| *Acanthochromis polyacanthus* | ACPOL | | * | *Siganus doliatus* | SIDOL | | * |
| *Amphiprion akallopisos* | AMALK | * | | *Siganus vulpinus* | SIVUL | | * |
| *Amblyglyphidodon curacao* | AGCUR | | * | *Scolopsis bilineatus* | SCBI | | * |
| *Chromis viridis* | CHVIR | * | * | *Scolopsis margaritifer* | SCMAR | | * |
| *Chromis weberi* | CHWEB | * | | *Scolopsis ciliatus* | SCCIL | * | |
| *Cheiloprion labiatus* | CLLAB | | * | *Parupeneus barberinoides* | PABAR | * | |

Appendix 6.2. **Relative abundance of fish species surveyed during the habitat degradation experiment at A) Maiton Island and B) Orpheus Island. Fish were listed according to their relative occurrence with species below the arrow occurring more than 10% of the time**

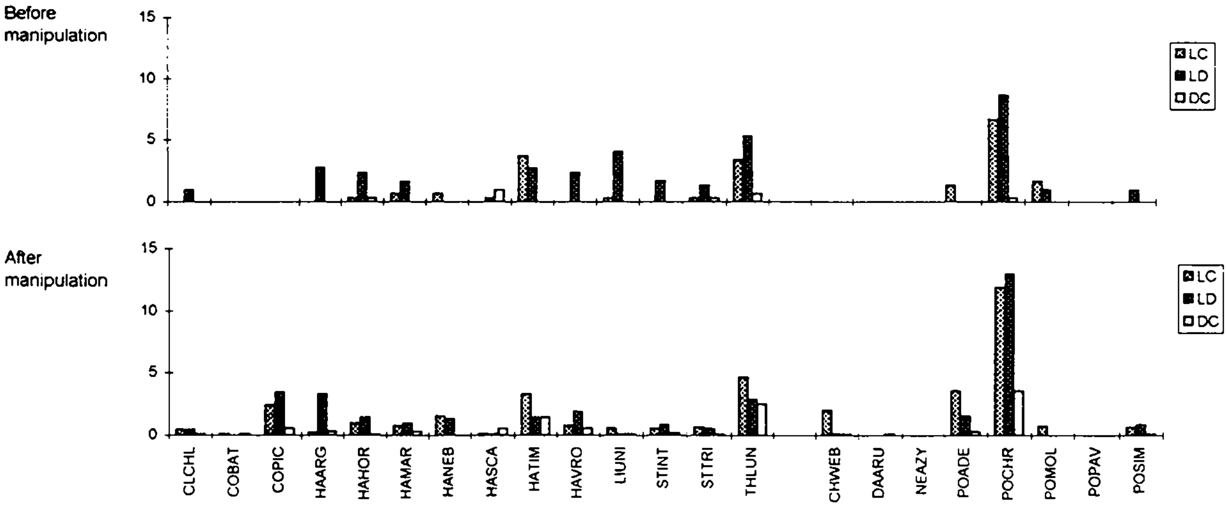


Appendix 6.3. Mean fish abundance (no. fish/25 sq. m) observed during the living coral degradation experiment at Maiton Island (Phuket, Thailand). LC = Living coral control, LD = Living coral degradation, DC = Dead coral control.

A) NE-Reef

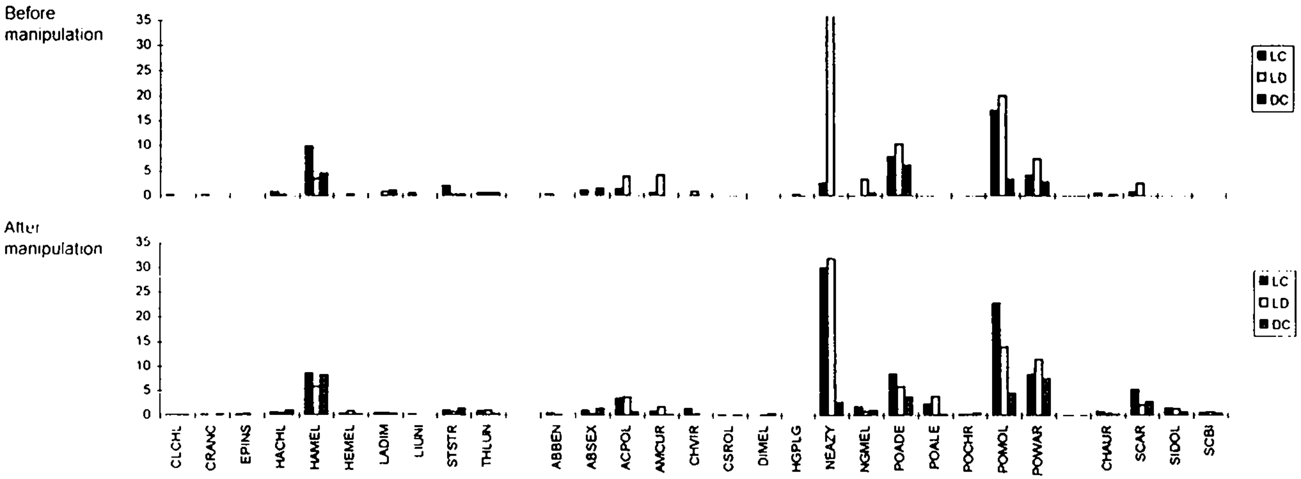


B) SE-Reef

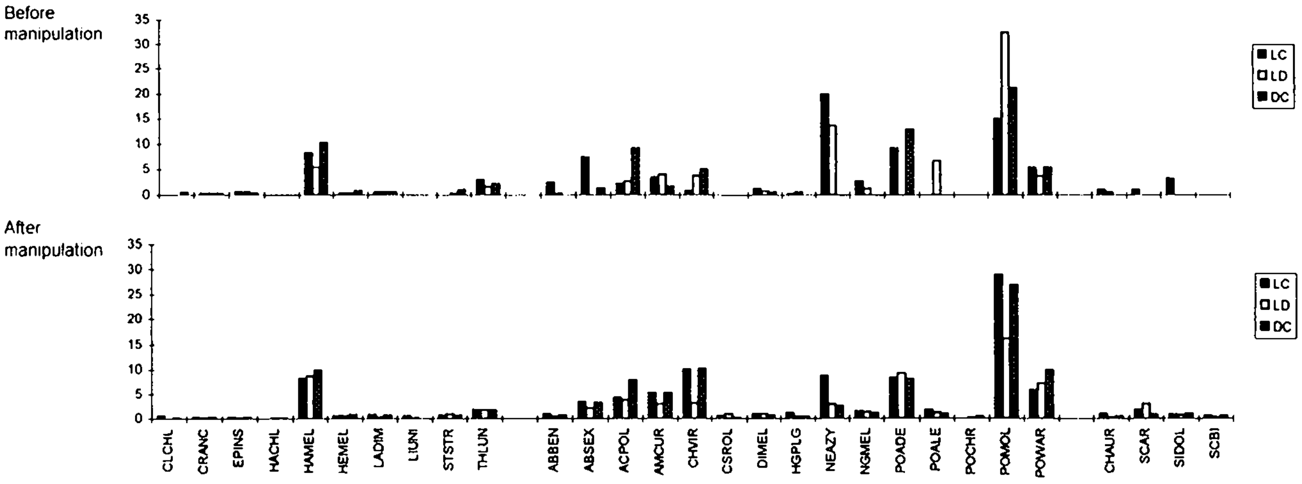


Appendix 6.4. Mean fish abundance (no. fish/25 sq. m) observed during the living coral degradation experiment at Orpheus Island (central GBR, Australia). LC = Living coral control, LD = Living coral degradation, DC = Dead coral control.

A) Pioneer Bay

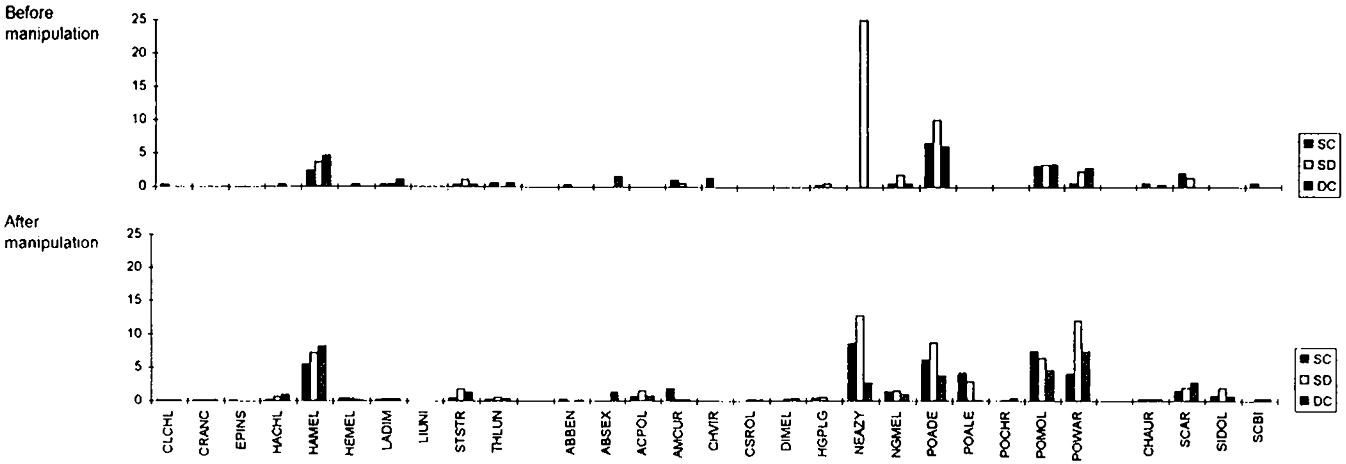


B) Cattle Bay

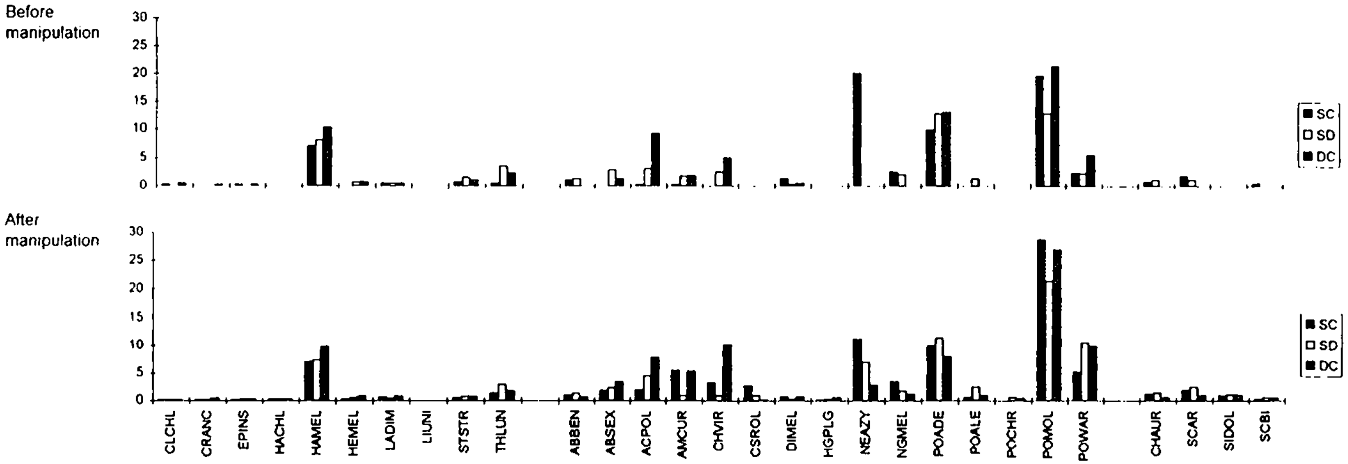


Appendix 6.5. Mean fish abundance (no. fish/25 sq. m) observed during the soft coral degradation experiment at Orpheus Island (central GBR, Australia). SC = Soft coral control, LD = Soft coral degradation, DC = Dead coral control.

A) Pioneer Bay



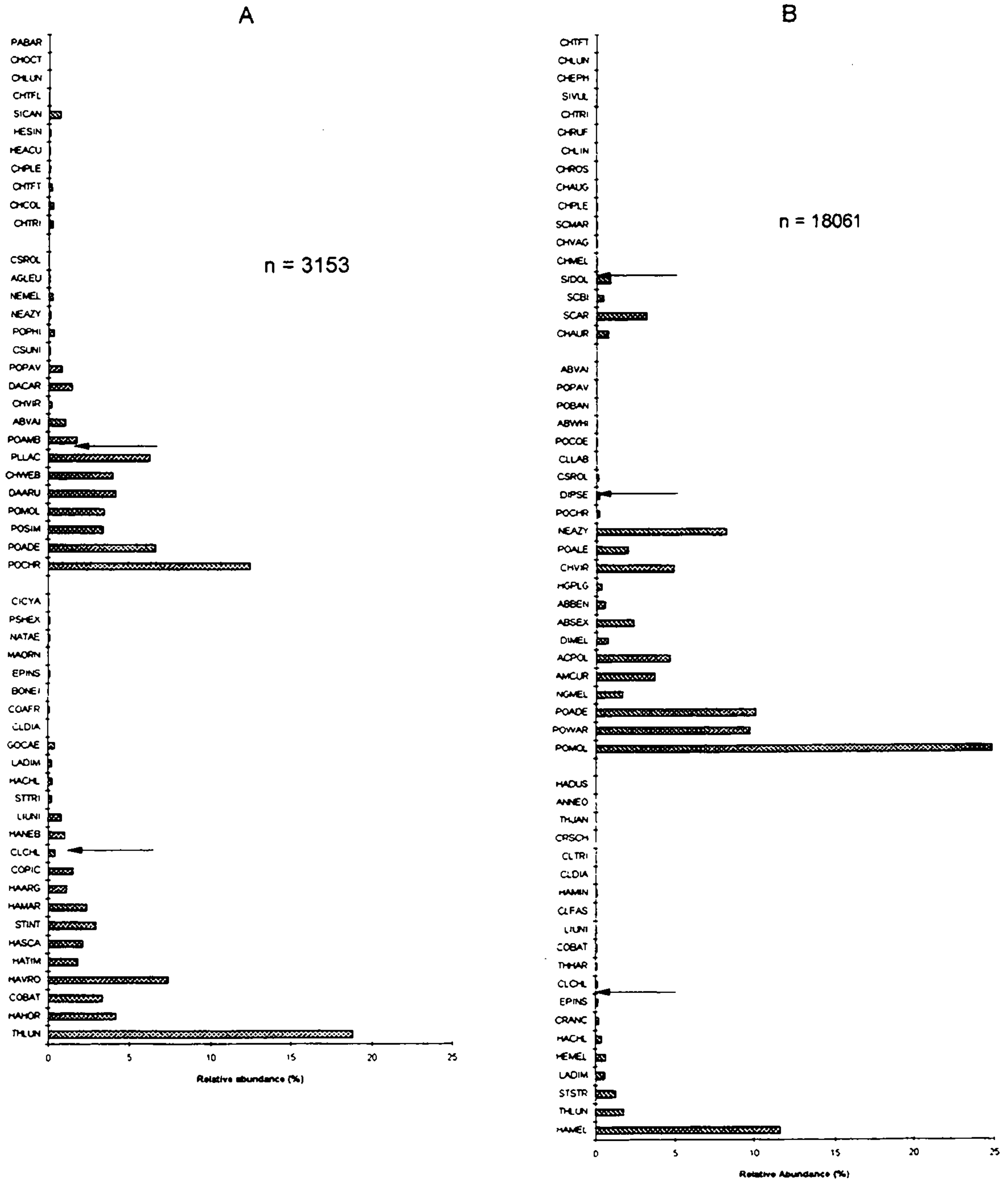
B) Cattle Bay



Appendix 7.1. List of coral reef fishes studied during habitat rehabilitation and enhancement experiment at Maiton Island (Phuket, Thailand) and Orpheus Island (central Great Barrier Reef, Australia).

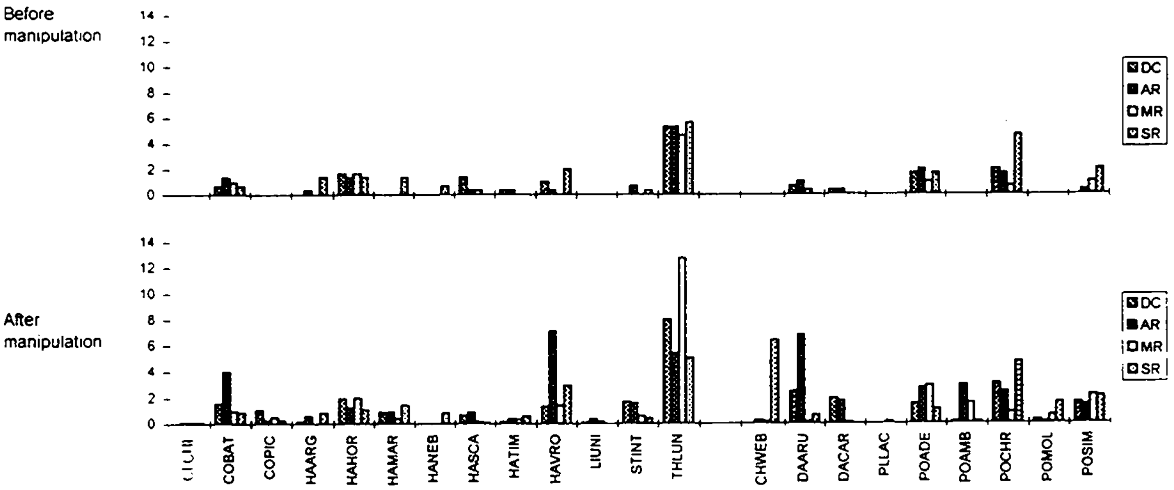
| Species | abbreviation | Maiton Island | Orpheus Island | Species | abbreviation | Maiton Island | Orpheus Island |
|---|---|---|---|---|---|---|---|
| LABRIDAE | | | | | | | |
| *Anampses caeruleopunctatus* | ANCAE | • | | *Chromis weberi* | CHWEB | • | |
| *Anampses meleagrides* | ANMEL | • | | *Cheiloprion labiatus* | CLLAB | | • |
| *Anampses neoguinaicus* | ANNEO | | • | *Chrysiptera rollandi* | CSROL | • | • |
| *Bodianus neilli* | BONEI | • | | *Chrysiptera unimaculata* | CSUNI | • | |
| *Cirrhilabrus cyanopleura* | CICYA | • | | *Dascyllus aruanus* | DAARU | • | |
| *Cheilinus chlorourus* | CLCHL | • | • | *Dascyllus carneus* | DACAR | • | |
| *Cheilinus digrammus* | CLDIA | • | • | *Dischistodus melanotus* | DIMEL | | • |
| *Cheilinus fasciatus* | CLFAS | | • | *Dischistodus perspicillatus* | DIPER | • | |
| *Cheilinus trilobatus* | CLTRI | | • | *Dischistodus pseudochrysopoecilus* | DIPSE | | • |
| *Coris africana* | COAFR | • | | *Hemiglyphidodon plagiometopon* | HGPLG | | • |
| *Coris batuensis* | COBAT | • | • | *Neopomacentrus azysron* | NEAZY | • | • |
| *Choerodon anchorago* | CRANC | | • | *Neoglyphidodon melas* | NGMEL | • | • |
| *Choerodon schoenleinii* | CRSCH | | • | *Plectroglyphidodon lacrymatus* | PLLAC | • | |
| *Coris pictoides* | COPIC | • | | *Pomacentrus adelus* | POADE | • | • |
| *Epibulus insidiator* | EPINS | • | • | *Pomacentrus alexanderae* | POALE | | • |
| *Gomphosus caeruleus* | GOCAE | • | | *Pomacentrus amboinensis* | POAMB | • | |
| *Halichoeres argus* | HAARG | • | | *Pomacentrus bankanensis* | POBAN | | • |
| *Halichoeres chloropterus* | HACHL | • | • | *Pomacentrus coelestis* | POCOE | | • |
| *Halichoeres dussumeri* | HADUS | | • | *Pomacentrus chrysurus* | POCHR | • | • |
| *Halichoeres hortulanus* | HAHOR | • | | *Pomacentrus moluccensis* | POMOL | • | • |
| *Halichoeres marginatus* | HAMAR | • | | *Pomacentrus pavo* | POPAV | • | • |
| *Halichoeres melanurus* | HAMEL | | • | *Pomacentrus philippinus* | POPHI | • | |
| *Halichoeres miniatus* | HAMIN | | • | *Pomacentrus similis* | POSIM | • | |
| *Halichoeres nebulosus* | HANEB | • | | *Pomacentrus wardi* | POWAR | | • |
| *Halichoeres scapularis* | HASCA | • | | OTHER MAJOR FAMILIES | | | |
| *Halichoeres timorensis* | HATIM | • | | *Chaetodon auriga* | CHAUG | | • |
| *Halichoeres vrolikii* | HAVRO | • | | *Chaetodon aureofasciatus* | CHAUR | | • |
| *Hemigymnus melapterus* | HEMEL | | • | *Chaetodon collare* | CHCOL | • | |
| *Labroides dimidiatus* | LADIM | • | • | *Chaetodon lunula* | CHLUN | • | • |
| *Labrichthys unilineatus* | LIUNI | • | • | *Chaetodon melannotus* | CHMEL | | • |
| *Macropharyngodon ornatus* | MAORN | • | | *Chaetodon ephippium* | CHEPH | | • |
| *Novaculichthys taeniourus* | NATAE | • | | *Chaetodon lineolatus* | CHLIN | | • |
| *Pseudocheilinus hexataenia* | PSHEX | • | | *Chaetodon octofasciatus* | CHOCT | • | |
| *Stethojulis interrupta* | STINT | • | | *Chaetodon plebeius* | CHPLE | • | • |
| *Stethojulis trilineata* | STTRI | • | | *Chaetodon rafflesi* | CHRAF | | • |
| *Stethojulis strigiventer* | STSTR | | • | *Chaetodon trifasciatus* | CHTFT | • | • |
| *Thalassoma hardwicke* | THHAR | | • | *Chaetodon trifascialis* | CHTFL | • | |
| *Thalassoma jansenii* | THJAN | | • | *Chaetodon triangulum* | CHTRI | • | • |
| *Thalassoma lunare* | THLUN | • | • | *Chaetodon vagabundus* | CHVAG | | • |
| POMACENTRIDAE | | | | *Chelmon rostratus* | CHROS | | • |
| *Abudefduf bengalensis* | ABBEN | | • | *Heniochus acuminatus* | HEACU | • | |
| *Abudefduf sexfasciatus* | ABSEX | | • | *Heniochus singularius* | HESIN | • | |
| *Abudefduf vaigiensis* | ABVAI | • | • | *Scarus juvenile* | SCAR | | • |
| *Abudefduf whitleyi* | ABWHI | | • | *Siganus canaliculatus* | SICAN | • | |
| *Acanthochromis polyacanthus* | ACPOL | | • | *Siganus doliatus* | SIDOL | | • |
| *Amphiprion akallopisos* | AMALK | • | | *Siganus vulpinus* | SIVUL | | • |
| *Amblyglyphidodon curacao* | AGCUR | | • | *Scolopsis bilineatus* | SCBI | | • |
| *Amblyglyphidodon leucogaster* | AGLEU | • | | *Scolopsis margaritifer* | SCMAR | | • |
| *Chromis viridis* | CHVIR | • | • | *Parupeneus barberinoides* | PABAR | • | |

Appendix 7.2. Relative abundance of fish during habitat rehabilitation experiments at A) Maiton Island (Phuket, Thailand) and B) Orpheus Island (central GBR, Australia). Fishes were listed according to their their relative occurrence, with species below the arrow occurring more than 10% of the time.

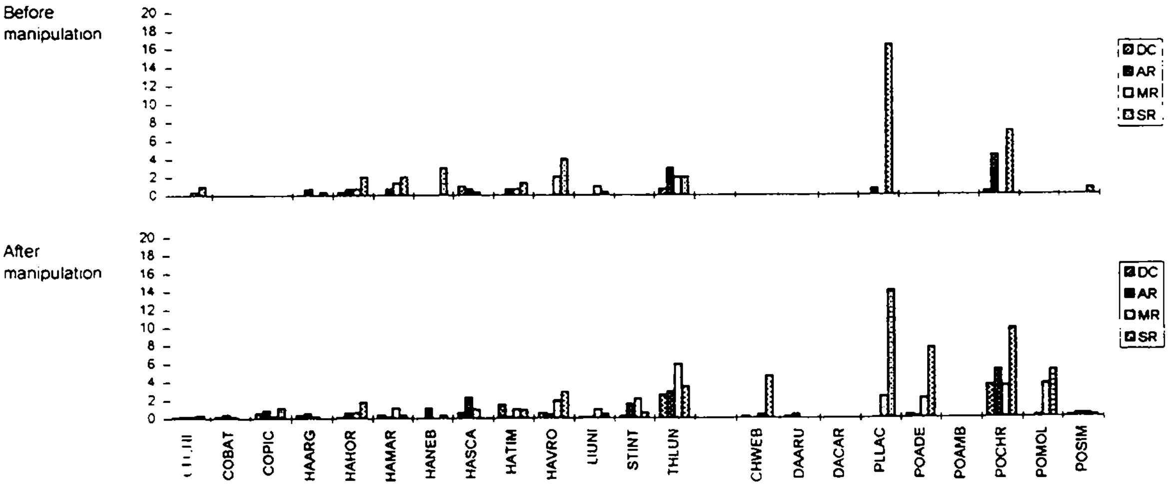


Appendix 7.3. Mean abundance (no. fish/25 sq. m) of fishes during the habitat rehabilitation experiments at Maiton Island.

A) NE-Reef

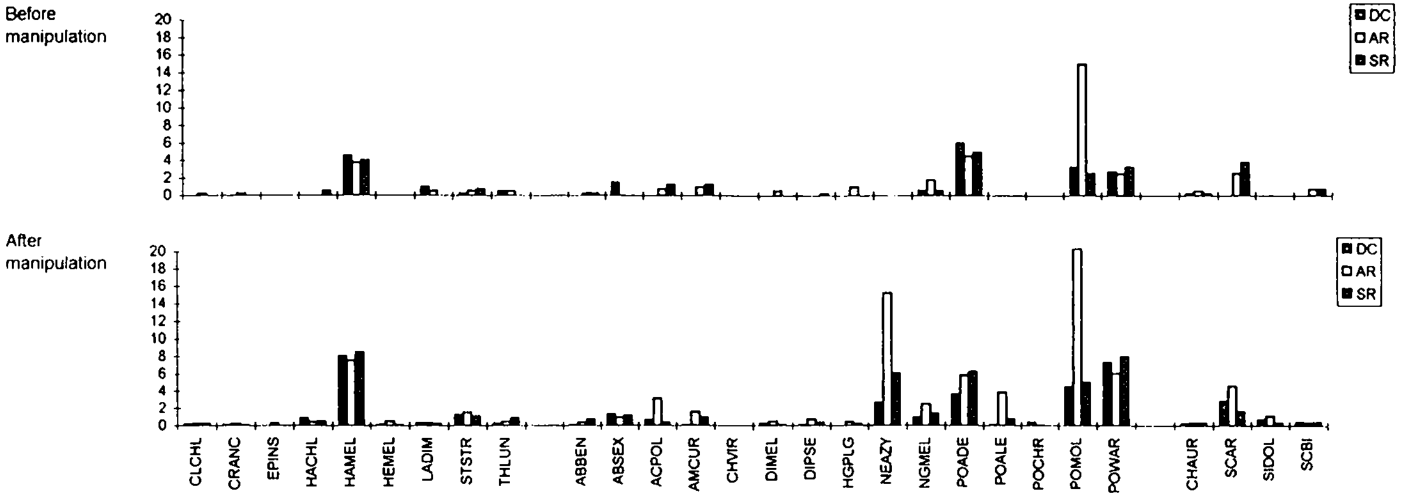


B) SE-Reef



Appendix 7.4. Mean abundance (no. fish/25 sq. m) of fishes during the habitat rehabilitation experiments at Orpheus Island.

A) Pioneer Bay



B) Cattle Bay