INTRODUCTION

Increasing concern about the ethical and economic trade-offs between exploitation and conservation of natural biological resources in recent years has placed greater pressure on environmental managers. The Great Barrier Reef is a particularly topical case, both within Australia and world-wide. The gazetting of the Great Barrier Reef as a multi-use marine park explicitly demanded the conservation of the biological characteristics of the Great Barrier Reef in the context of ongoing recreational use and commercial development (GBR Marine Park Act, 1975). As managers of the GBRMP, therefore, the Great Barrier Reef Marine Park Authority (GBRMPA) is faced with balancing the interests of an array of commercial activities (*e.g.* tourist industries; scale, crustacean and sessile invertebrate fisheries; commercial shipping), recreational users, Aboriginal users, and research groups, whilst endeavouring to ensure that the bio-physical system is conserved. Management in this context effectively entails the regulation of human use, *eg.* by zoning areas for different allowable uses, rather than intervention in natural bio-physical processes *per se* (Hendee *et al*, 1990). The success of such regulatory management practices should be assessed with reference to two variables: i) status of the ecosystem that is to be conserved; and ii) the degree to which human use is facilitated to the satisfaction of users. Monitoring the status and dynamics of the natural system, and of human use of the system, therefore, provides the feedback necessary to assess the success or failure of management strategies. The quality and quantity of information derived from such monitoring is crucial to the evolution of prudent and justifiable management (Hendee *et al*, 1990).

Management responsibilities in the GBR region are manifest at a variety of scales of space and time. Impact assessment and issues of reef use are typically addressed at relatively local scales (within reefs) and over short times (one to five years). Zoning of the GBR and general management strategies, however, extend to very large spatial scales (reefs, regions) and are operative over long times (5 years - decades). To adequately assess the responses of reef biota to these multiple activities requires sound empirical knowledge of spatial and temporal patterns in the distribution and abundance of organisms on the GBR under 'normal' conditions, and of the resilience of populations to perturbation. This information is most efficiently provided by carefully planned quantitative descriptive studies over a range of spatial and temporal scales - i.e., *via* a sound monitoring programme - supplemented with the manipulative experimental studies that are the province of fundamental research.

The information derived from monitoring is, in part, likely to be of a general, descriptive nature intended to provide an empirical context within which to make decisions about the management of specific issues or perturbations, and within which the results of local fundamental research can be better interpreted. It is important therefore, that field sampling is designed carefully to provide context-information for a variety of issues, as well as providing empirical tests of the effects of management strategies on biota (Alcala 1988, Craik 1981, Crimp 1986, GBRMPA 1978, 1979, Hendee *et al* 1990, Russ 1984, 1989). In most cases, management decisions will be made on the basis of existing information, such as that delivered by general monitoring, or after relatively short term studies. Typically it will be desirable to integrate information from different sources in a coherent way. The strongest and most general basis for coherent integration among studies is sound empirical knowledge about the sampling and analytical procedures from which each piece of information was derived (Andrew & Mapstone 1987, Green 1979). The choice of sampling methods for general monitoring, therefore, should be related to methods used widely in other studies, and the documentation of their characteristics will be an important component in the development of monitoring programmes.

The implementation of a general monitoring programme over such a range of scales will be expensive and logistically constrained. It is critical, therefore, that sampling methods are cost-effective. That is, methods should be chosen that i) provide reliable data ii) will maximise the potential to identify changes or patterns in the abundance of biota; iii) are logistically feasible to

repeatedly implement in a wide range of situations; and iv) are cheap, so that maximum flexibility for the design of monitoring can be retained, within what will always be limited budgets.

In addition to the above, mainly logistic considerations, the final design of a general monitoring programme must be based on considerations of the biology of the organisms of interest, the ways in which we perceive and count them, and the scales at which natural fluctuations in abundances occur and management actions are likely (Andrew & Mapstone 1987, Green 1979, Resh 1979). Particularly important biological considerations include:

- the spatial and temporal characteristics of dispersion of the organisms;
- the rates and ranges of movement of the organisms, particularly with respect to the size of sampling units and the time taken to count them;
- the spatial and temporal scales at which abundances vary most.

Important methodological considerations are:

- the size and shape of unit within which organisms are counted;
- the rates and ways in which the units are surveyed by an observer;

Critical aspects of the design of a monitoring programme are:

- the numbers of units used;
- the arrangement of sampling units at spatial and temporal scales greater than those of the sampling units.

Interactions among the above biological, methodological, and design factors will determine the degree to which real patterns in abundances are reflected in our data, the precision of estimates obtained from a sampling programme, and the resolving power of statistical tests based on those estimates. The most effective allocation of available resources rests on a thorough understanding of these factors.

Thus, an early step in the development of a monitoring programme should be the examination of the relationships between methodology and small-scale biological features, resulting in the choice of sampling unit and method of survey that provides an optimum balance between reducing the costs of local sampling and maximising the precision of estimates for each subject species or group of organism. It should be verified, as far as possible, that the chosen sampling unit has adequate sampling characteristics over the range of environmental conditions (*e.g.* habitat, population density) within which it will be used. Non-destructive methods are generally preferred to methods that alter the environment being monitored. Subsequently, the optimum sampling method is used to estimate the variation in abundances at a range of scales, and these estimates are then employed to decide where best to allocate limited resources to answer a specific question.

Assuming that a range of sampling methods are feasible within an envelope of cost and logistic considerations, two main empirical features have been considered to distinguish which sampling units and/or methods are most desirable: i) accuracy or relative bias; and ii) precision. The accuracy of an estimate refers to its degree of departure from the true population value (Andrew & Mapstone 1987, Lincoln *et al* 1982, Sokal & Rohlf 1981, Underwood 1981). Bias refers to the consistency of inaccuracy - *ie* the extent to which repeated estimates tend to differ from the 'truth' in the same direction. Both accuracy and bias will be extremely difficult to asses in absolute terms for field studies because we generally always have only estimates of the parameter of interest and, therefore, can never be sure of its true value. The relative bias of a number of estimates can be inferred, however, if it can be assumed that all are biased in the same direction - *i.e.*, if it is probable that all are either over estimates or underestimates. In general, sampling units that provide data with the smallest bias will be preferred. The precision of an estimate refers to the expected variation in repeated estimates of the same population (Andrew & Mapstone 1987, Cochran 1963, Cochran & Cox 1957, Elliot 1977, Lincoln *et al* 1982, Sokal & Rohlf 1981), and is often expressed as the (unit-less) ratio of standard error to mean. The precision of an estimate is independent of its bias, and is by definition a relative property of a estimate unrelated to any absolute 'truth'. Hence, comparisons of precision are straightforward. Methods which provide better precision for the same sampling effort are preferred.

Several authors have found that either or both of these properties vary greatly with sampling method (*e.g.* Fowler 1987, Gray and Bell 1985, Lincoln Smith 1988, 1989, Samoilys & Carlos 1992), and/or the size of sampling units (see review by Andrew & Mapstone 1987, Downing & Anderson 1985, Downing & Cyr 1985), and may vary among habitats, times, sites etc. for a given sampling unit (Short & Bayliss 1985). These and other authors have raised the spectre of spurious inferences arising from inappropriate choice of sampling methods, and have emphasised the need to quantify the characteristics of precision and bias associated with chosen methods (Bros & Cowell 1987, Green 1979, Downing 1979, Pringle 1985).

The sampling characteristics of several methods of visual survey have been examined previously (*e.g.* hectare counts of *Plectropomus* spp. (Ayling 1983a, Ayling & Ayling 1984a, GBRMPA 1979); timed visual surveys and/or belt transects for estimating abundances of fish (Bohnsack & Banerot 1983, DeMartini & Roberts 1982, GBRMPA 1978, 1979, Kimmel 1985, Samoilys & Carlos 1992, Sanderson & Salonsky 1980); manta-tow, and strip-transect counts of holothurians over sand (Harriott 1984); manta-tow surveys of hard corals and *Acanthaster planci* (Fernandez, 1990, Fernandez *et al* 1990, Kenchington 1978), line-intercept methods, point, and quadrat methods (Weinberg 1981), and video methods (AIMS, Mapstone in progress) for estimating abundance of sessile fauna. Whilst the manta tow technique has received considerable scrutiny, there continues to be some uncertainty over the strengths and weaknesses of methods for the survey of benthic biota and demersal fishes on the GBR (Ayling 1983a, Ayling & Ayling 1984a, Crimp 1986, 1987, GBRMPA 1978, 1979, 1986),). In most cases, the relative bias of survey methods have been the primary characteristic of interest, in many cases based on a desire to adequately characterise the assemblage structure of the sampled areas (e.g. GBRMPA 1978, Russell *et al* 1978, Sale & Douglas 1981). Considerations of the precision of estimates of abundances or the utility of methods in the context of the likely design characteristics of monitoring programmes, however, have received less attention.

In this study we investigated the methodological aspects of estimating the abundances of a number of reef organisms. We were concerned primarily with the utility of belt (or strip) transects of various sizes as sampling units for the estimation of population densities. The characteristics of relative bias and precision of sampling by belt transects have not been considered in the context of a multi-species monitoring programme, although their application to counting particular species have been examined previously (Ayling 1983a, Ayling & Ayling 1984a, Fowler 1987, Mapstone 1988, Sale & Sharp 1983). The organisms with which we were concerned were relatively large, discrete organisms amenable to rapid counting in the field by divers, which were of interest to the GBRMPA as measures of management success (GBRMPA 1978), and which had been surveyed by related methods in the past (*e.g.*, Ayling 1983b,c, Ayling & Ayling 1984b,c, 1985, 1986a,b, Doherty 1987, Doherty & Williams, 1988, Fowler 1987, Mapstone 1988). Our interest was in the selection of optimum methods for the estimation of population densities in the framework of future monitoring programmes for the GBR, and we were not considering the utility of visual survey as a means of estimating population structure (see Ayling 1983b,c, Ayling & Ayling 1984b,c, 1985, 1986a,b, 1991, 1992a,b, Ayling et al 1991, Crimp 1987, GBRMPA 1978, 1979).

Thus, the project had two primary objectives.

1. Estimation of relative biases caused by sampling unit dimensions, diver activity, different observers, and searching procedures.
2. Estimation of the characteristics of precision of estimates from units of different size and the relative cost-efficiency of sampling with those units. Relative cost-efficiency here means the expenditure required to obtain estimates with uniform precision.